THE MINERALISATION AND FATE OF NITROGEN FOLLOWING THE INCORPORATION OF GRASS AND GRASS-CLOVER SWARDS

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DECLARATION

I, Mark Geoffrey Davies, declare that this thesis was composed by myself, and the work described was carried out by myself, except for the instances detailed in the text and acknowledgements.

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ABSTRACT

Field and laboratory experiments were carried out between 1992 and 1994 to study the release and fate of nitrogen (N) following the incorporation of grass and grass-clover swards. The effects of sward type, quality of sward residue and previous grazing management on the N cycle were studied. Techniques for the measurement of mineralisation and nitrate (NO_3^-) leaching on medium textured soils were also assessed.

Initial work on the measurement of mineralisation using ^{15}N pool dilution techniques revealed major problems with practical application in the field. Mineral N release was therefore estimated by combining soil mineral N, NO_3^- leaching, plant uptake and gaseous N loss data.

Evaluation of estimated leaching loads and specific tests comparing porous cups with three other estimates of soil water NO_3^--N concentrations cast serious doubts on the quantitative accuracy of porous cups on medium textured soils. Results suggest that cups were sampling relatively immobile water and, depending on the source of the solute, this led to over- and under-estimates of leaching loads.

The ploughing out of previously grazed swards released a substantial amount of N in the first eighteen months following incorporation (*ca.* 370 kg N ha^{-1}), particularly during the first two months. Swards subject to an unfertilised cutting regime released much less N following ploughing. This was attributed to a reduction of soil organic matter (SOM) "degradability" and, to a lesser extent, plant residue N supply. Clover residues were found to have a notable effect on mineral N release only when sward clover contents were over 20% of total dry matter (DM) and grass residues were of much lower N content.

The ploughing out of grassland produces a considerable short term increase in nitrous oxide (N_2O) emissions thought to be initiated by the intimate mixing of readily available carbon sources into the soil during rotavation. The influence of clover residues on N_2O emissions was greater than their influence on mineral N release patterns.

Manipulation of the post-ploughing N cycle may be possible by altering sward management prior to ploughing. The cessation of grazing for a short period before

sward incorporation could reduce mineral N release and N_2O emissions. However, the capacity to reduce N_2O emissions is constrained by weather conditions at critical times following incorporation.

CONTENTS

LIST OF ABBREVIATIONS AND SYMBOLS

Br = bromide
C = carbon
CO_2 = carbon dioxide
C_2H_2 = acetylene
d.f. = degrees of freedom
DBD = dry bulk density
DM = dry matter
MIT = mineralisation immobilisation turnover
MOM = macro-organic matter
N = nitrogen
N_2O = nitrous oxide
NH_4^+-N = ammonium-nitrogen
NO_3^--N = nitrate-nitrogen
O_2 = oxygen
SE = standard error of the mean
SOC = soluble organic carbon
SOM = soil organic matter
WFPS = water-filled pore space

1 INTRODUCTION

Along with Ireland, Britain has the most favourable climate in Europe for providing grassland products (Newton, 1993). Grassland, which can be cut or grazed, may be part of a ley-arable rotation or a more permanent grassland system. Occasionally both types of grassland need to be ploughed up to allow for arable crop production and sward reseeding, respectively. Soil nitrogen accumulation under grassland and its subsequent release following ploughing is an essential element in low input agricultural systems. However, the disturbance of the soil system can also result in considerable loss of nitrogen (N) from the soil and has been associated with the contamination of groundwater supplies by nitrate (NO_3^-) (Cameron and Wild, 1984).

The amount of N released following the ploughing out of grassland depends upon sward composition, age and previous management. The impact of the grazing animal on the grassland soil N cycle has become a focus for recent research. This project aimed to investigate the release of mineral N following the ploughing out of clover-rich and grass-dominated swards, subject to cutting or grazing regimes.

The ability of clover to fix atmospheric N could play an increasingly important role in modern agricultural rotations. The efficient use of this 'fixed' N as a substitute for part, if not all, artificial fertiliser N requires an understanding of the soil N cycle following ploughing out of clover-rich swards.

Unlike fertiliser N, N in plant and animal residues, and soil organic matter (SOM), must be mineralised from its organic form into ammonium (NH_4^+) and NO_3^- before it is available to following crops. Nitrate is readily lost by leaching in soil solution beyond the root zone and, during anaerobic conditions, may be lost as nitrogenous gases. These processes represent an economic loss to the farmer. In addition, these losses add to the environmental problems of NO_3^- pollution of water supply and accumulation of atmospheric greenhouse gases, respectively. Hence, the key objectives of this research were to:

a) Quantify the amount and timing of the release of mineral N from ploughed out clover-rich swards during the first two years after cultivation.

b) Study the fate of this mineral N during the first two years after cultivation, quantifying:

i) potential plant uptake;

ii) gaseous losses of N;

iii) leaching losses of N.

c) Assess the impact of sward composition and sward management on the soil N cycle following incorporation of grassland.

These objectives were met through the establishment of three field trials, and a subsequent incubation experiment, allowing analysis of the net effect of disturbance of the grassland system, the net effect of sward clover content and, through a change in sward management, the effects of grazing on the soil N cycle following ploughing.

Research was also carried out to assess the use of ^{15}N for the measurement of gross N mineralisation (Appendix 6). Additional work was carried out to evaluate the appropriateness of various techniques available for the measurement of NO_3^- leaching on medium textured soils.

2 LITERATURE REVIEW

2.1 GRASSLAND SYSTEMS

2.1.1 Grassland in the United Kingdom

The area of agricultural land in the United Kingdom (UK) is 18.4 million ha, representing 77% of the total land area. Grassland makes up about 12 million ha or 65% of the agricultural land area (Ministry of Agriculture, Fisheries and Food (MAFF), 1996a). The total area under temporary and permanent grassland on the lowlands has always considerably exceeded the total arable area, despite large reductions in grassland since the 1930's (Holmes, 1989). The percentage of agricultural land under grass tends to increase from east to west and from south to north as a result of increased rainfall, generally decreasing soil fertility and the difficulties of arable cultivation (Frame, 1992a).

Grassland is categorised into swards under 5 years old, over 5 years old and rough grazing. Grassland over 5 years old includes temporary grassland or leys, which may be part of an arable-grass rotation and possibly up to 10 or more years old, and permanent grassland (Frame, 1992a). The term permanent grassland describes land maintained as grassland without the intervention of ploughing and reseeding (Holmes, 1989). In 1995 there were 1.4 million ha of grassland under 5 years and 5.3 million ha over 5 years old (MAFF, 1996a). A survey of grassland in England and Wales indicated that of the grassland 5 years old and over, 23% was from 5 to 8 years old, 22% was 9 to 20 years old and 55% was more than 20 years old (Green and Williams, 1975).

2.1.2 The importance of nitrogen

Nitrogen is the key element which drives production of temperate grassland systems, accounting for at least half of the output and much of the profit (Garrett *et al.*, 1992). The average use of fertiliser on grassland in Scotland, excluding rough grazing, has been fairly stable in recent years (MAFF, 1996b). In 1995, 143 kg N ha^{-1} was applied to grassland under 5 years old, whilst 100 kg N ha^{-1} was applied to grassland over 5 years old (MAFF, 1996b), although figures range from 0 to 450 kg N ha^{-1} (Frame, 1992b). About half of the 1.5 million tonnes of fertiliser N used annually in Britain is applied to grassland (Frame, 1992b).

2.1.3 Nitrogen fixation and clover potential

The importance of leguminous plants lies in their rhizobial N-fixation ability, soil improvement properties and excellent feeding value for animal production. These advantages have been neglected in Europe in recent times (Frame, 1992c) and reliance on white clover is vastly below its potential (Hopkins *et al.*, 1994). This has been due to several shortcomings of forage legumes: increased unpredictability; lower spring yields; poor persistence; potential to cause bloat; small legume seeds are harder to establish (Frame and Newbould, 1986; Laidlaw and Frame, 1988). Farmers are not always able to maintain the optimal grazing and cutting regime for clover swards and they use too much fertiliser N to compensate for poor spring growth (Frame and Newbould, 1986). However, present extensification objectives in European Union (EU) countries have caused renewed research interest in grass-legume swards (Parsons *et al.*, 1991; Frame, 1992c).

Whilst there is no up-to-date survey of white clover use in the UK, Hopkins *et al.* (1994) concluded that 15-30% of the grassland used in sheep production contains white clover, but only 10-15% contained a significant amount (>25% white clover).

White clover is a perennial legume which spreads through the sward by means of branching stolons, and is by far the most important forage legume used in Britain. The N-fixing bacteria of the root nodules are concentrated in the upper sward layers. In comparison with grass, clover is richer in protein and minerals but lower in fibre. While best suited as a grazing plant, it can also be successfully cropped for high quality silage (Frame, 1992c).

Garrett *et al.* (1992) reported N fixation in grass-clover swards of 167-236 kg N ha^{-1} yr^{-1} during a 3 year trial in Northern Ireland, whilst lower rates, 100-150 kg N ha^{-1} yr^{-1}, may be expected on hill and upland pastures (Frame, 1992b). An average amount fixed for a sward in which white clover is well established would be about 150 kg N ha^{-1} yr^{-1}. In practice, a target for sward clover content would be 25-35% of the annual herbage dry matter (DM) (Frame, 1992b). At a nil rate of applied N, an all-grass sward produces 2-5 t DM ha^{-1} yr^{-1}, depending on the available N in the soil, whilst a grass-white clover sward produces 6-9 t DM ha^{-1} yr^{-1}, the higher level being associated with 25-35% of N-fixing clover (Frame, 1992b). Ball (1979) found that about 80% of the N fixed by white clover in an unfertilised grass-clover sward was recovered in herbage. Under most conditions N is principally transferred to the

grass component above-ground through the decomposition of legume leaf litter and green leaf trampled into the ground by grazing animals and through excretal returns from animals grazing the legumes (Henzell and Vallis, 1977; Vallis, 1978). Below-ground transfer can occur when roots and nodules die back following heavy defoliation or death of the legume (Vallis, 1978). Ledgard (1991) estimated that 70 kg N ha^{-1} of the 269 kg N ha^{-1} yr^{-1} fixed was transferred to grass in this way.

The use of leguminous crops to maintain the N status of soil provides a means of reducing the dependence on commercial N fertiliser in the production of food. As mineral fertilisers become more costly in terms of energy consumption, leguminous crops become a more viable resource for providing the available soil N necessary to maintain high crop yields (Frankenberger and Abdelmagid, 1985). For proper crop management, more information is needed about the factors affecting the release of N from nitrogenous plant materials (Müller *et al.*, 1988). The acquisition of such information would be aided by the development of a reference method which is known to measure mineralisation rates accurately under field conditions (Raison *et al.*, 1987).

2.2 SOIL NITROGEN CYCLE

The soil N cycle is shown in Figure 2.2. The pools and processes involved in this cycle are described in detail in the sections below.

2.2.1 Soil organic nitrogen

Over 90% of the N in the surface layer of most soils is organically combined (Stevenson, 1982a). The total amount of N in many soils is appreciable, often exceeding 4000 kg N ha^{-1} to the depth of ploughing (Stevenson, 1982b). The importance of this organic N from the standpoint of soil fertility has long been recognised (Stevenson, 1982a). However, our knowledge of humic substances is incomplete, due primarily to their extremely complex structure (Haynes, 1986a).

Soil organic matter and organic N is composed of a continuum of organic materials stabilised to varying degrees against mineralisation by molecular recalcitrance, physical separation from the soil microbial biomass and/or direct association of the substrate with inorganic ions and clay surfaces (Skjemstad *et al.*, 1988). This continuum is divided into pools, defined as amounts of material behaving similarly

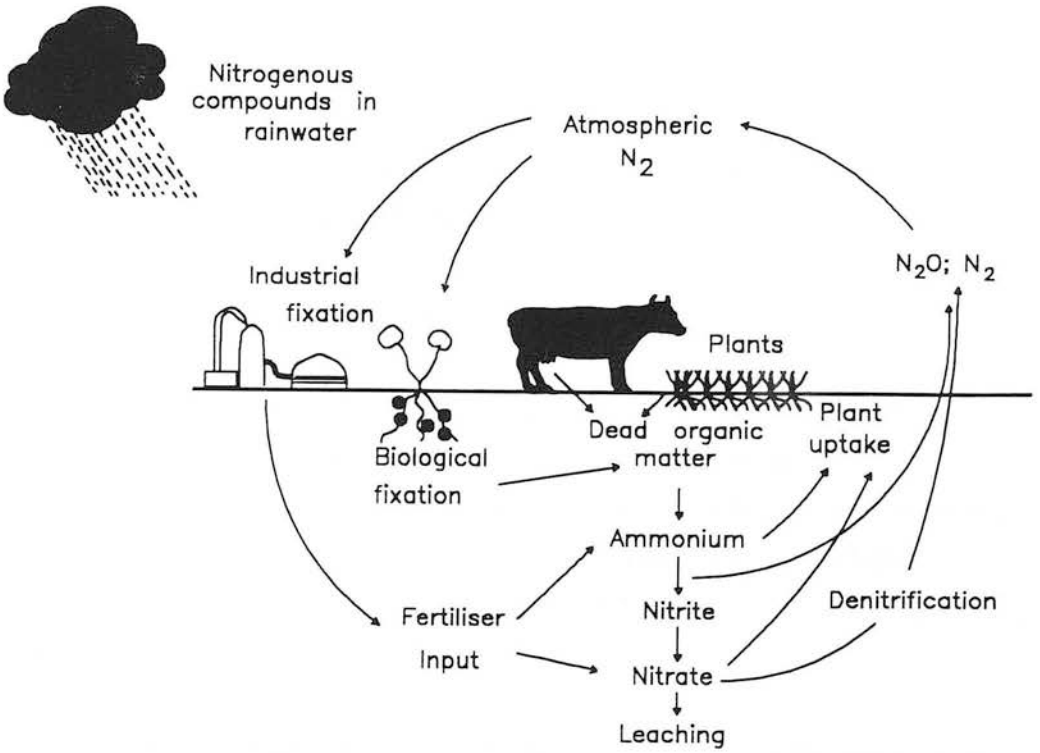


Figure 2.2 The soil nitrogen cycle (adapted from Haynes (1986a)).

enough to be grouped together (Bjarnason, 1989). The size of three biologically interesting phases (or pools) of SOM can be estimated, namely, the living biomass (by the fumigation method), the fresh debris making up the active phase (by isotope dilution calculations in short-term net mineralisation experiments, with correction for the separate biomass determination), and the very old and passive material (by ^{14}C (Carbon) dating) (Jansson and Persson, 1982). The biomass can contribute substantially to the pool of mobile, plant available nutrients in the soil (Paul, 1984). The quantity of N in the microbial biomass of an unmanured wheat field was 3.5% of the total soil N content (95 kg N ha^{-1}), with higher figures expected for grassland soils (Jenkinson and Ladd, 1981).

The size of the SOM pool tends towards an equilibrium at a diminishing rate over time (Jenny, 1941). Richardson (1938) estimated that the half period for grassland to reach equilibrium was 25 years, but according to Garwood *et al.* (1977) only 10-12 years of consistent management are required to reach equilibrium. This discrepancy may, at least in part, be because Richardson (1938) studied cut grassland whilst Garwood *et al.* (1977) studied grazed grassland under which SOM may be expected to accumulate faster (section 5.2). The SOM active fraction size and turnover rate

are related to agricultural practices and soil vegetation type (Paul, 1984). Legume-based pastures used in rotation with cereals do not necessarily increase the total organic matter content. However, the "light fraction" composed of predominantly finer, non-humified SOM and considered more active in nutrient supply, does increase (Russell, 1966; Greenland, 1971).

2.2.2 Mineralisation-immobilisation

The relatively small amount of N present in inorganic form as NH_4^+ and NO_3^- is significantly greater in agricultural soils, especially those under intensive management, than in other ecosystems (Vinten and Smith, 1993). In grassland soils mineral N accounts for <0.5% of the total N, when fixed NH_4^+ is excluded (Woodmansee *et al.*, 1981). Many processes affect the size of this pool (Figure 2.2), and it is subject to rapid turnover and change.

Mineralisation is the transformation of N from organic N into the inorganic forms NH_4^+ and NO_3^-. The process is carried out by heterotrophic soil organisms that utilise nitrogenous organic substances as an energy source (Jansson and Persson, 1982). Mineralisation may be considered as the driving force behind other soil N transformations, as it precedes plant uptake or losses from the soil of organic N (Rees, 1989). Immobilisation, the transformation of inorganic N compounds into organic compounds by assimilation and transformation in cells and tissues, works simultaneously and in the opposite direction to mineralisation (Jansson and Persson, 1982). Immobilisation has been shown to occur predominantly from the pool of NH_4^+ (Jansson, 1958; Recous *et al.*, 1988). However, where NH_4^+ is unavailable, microbial assimilation of NO_3^- will occur in the presence of readily available C (Azam *et al.*, 1986; Recous *et al.*, 1988). The net product of these processes depends upon their balance. Any N released is in excess of microbial demand (net mineralisation), whilst any N immobilised from the soil N pool is to make up for N deficits in the material being decomposed (net immobilisation).

These processes make up the heterotrophic subcycle, within the full N cycle. Jansson and Persson (1982) refer to this continuous transfer back and forth between organic and inorganic N forms, which underlies the build up and decay of the heterotrophic biomass, as mineralisation immobilisation turnover (MIT). This is an important concept because it stresses turnover and gross rates, rather than net outcomes, urging a fuller analysis of N transformations. For example, a measurement of low net

mineralisation does not reveal whether it is due to low general activity or high activity in opposite directions. This can be investigated using ^{15}N-labelled substrates, or by measuring carbon dioxide (CO_2) evolution to assess levels of microbial activity.

2.2.2.1 Factors affecting mineralisation

Environmental factors affect the speed of decomposition and release and therefore influence the short term dynamics of N mineralisation.

Soil Temperature

Mineralisation rates generally rise rapidly with increasing temperature, over the range normally found in soils in the field. The optimum temperature for organic matter decomposition is around 35°C (Bunt and Rovira, 1955; Alexander, 1977), predominantly due to the effects on microbial activity. Some physical effects exist, such as adsorption of NH_4^+ salts onto clay surfaces, which decreases with increasing temperature (Nõmmik, 1981). Below temperatures of about 2°C microbial activity is greatly reduced (Alexander, 1977). However, Müller and Sundman (1988) found there was still high N release from legume residues, despite soil being frozen for five out of six and a half months. The authors suggested this was because the easily soluble nature of the compounds available did not require much microbial degradation. Tam *et al.* (1983) showed rapid litter degradation at temperatures as low as 0°C.

In most soils there is an approximate doubling of microbiological activity for each 10°C rise in temperature between 5-35°C (Killham, 1994). This can be described as a ratio of the velocity coefficients at temperatures t°C and t+10°C, Q_{10}, which in this case is about 2.

Relationships developed in laboratories at constant temperatures may not necessarily reflect those of fluctuating field conditions. Stanford *et al.* (1975b) showed that temperature fluctuations between 5 and 35°C had no effect upon the amount of N mineralised and Biederbeck and Campbell (1971, 1973) came to similar conclusions for the range 3-14°C. However, fluctuations around the freezing point may produce more marked effects. Gasser (1958) found that frequent freeze-thaw cycles increased mineralisation rates, whilst a long period of freezing had little subsequent effects on mineralisation patterns. Biederbeck and Campbell (1971, 1973) proposed that

transient cold spells killed microbial cells which then serve as a readily available substrate for surviving microbes.

Soil Moisture

Microbial activity is highly dependent on moisture content, and hence so is mineralisation (Jenkinson, 1981). Too little moisture slows microbial activity, whilst too much water restricts oxygen (O_2) diffusion through the soil and thus inhibits aerobic decomposition. The rate of decomposition by aerobic bacteria is much greater than that brought about by anaerobic bacteria, because the former are more energy efficient (Vinten and Smith, 1993).

Between wilting point (-1500 kPa) and field capacity (-10 to -50 kPa) ammonification increases, but above and below these limits rates decrease (Reichman *et al.*, 1966; Stanford and Epstein, 1974). Stanford and Epstein (1974) found a strong linear relationship ($Y=-3.9+1.02X$; $r=0.93$) between relative N mineralisation and relative soil water content in the range 20 to 100% of the optimum water content for mineralisation. These results were for constant laboratory conditions and, just as for temperature, fluctuations in the field may change patterns. Drying, followed by rewetting, has been found to cause N flushes for one to two weeks (Birch, 1960) and increased N release from grass tissues added to an acid soil (Birch, 1964). In contrast to Birch (1964), van Schreven (1968) found that although drying stimulated the subsequent mineralisation of C and N from soil humus it retarded the mineralisation of fresh plant materials.

The size of the flush is positively related to the humus content, the dryness of the soil and the length of time the soil has remained dry. Hypotheses explaining the flush include N release from cells of the killed biomass (Black, 1968), a chemical change in organic matter due to drying (Waksman and Starkey, 1923) and swelling and shrinking of soil (Russell, 1966), all of which increase organic matter availability to decomposers. Marumoto *et al.* (1982) calculated that during the first 4 weeks following a drying and rewetting cycle about 40 kg N ha^{-1} in the upper 12.5 cm of soil was derived from microbial cells after death.

Soil Texture

Soil texture can affect mineralisation both indirectly and directly. Clay soils will hold more moisture and thus inhibit organic matter decomposition under wet conditions, by slowing O_2 diffusion through the soil. Directly, clays can protect organic matter by adsorption of molecules onto clay surfaces, thus decreasing availability to microorganisms, and enzymes carrying out degradation may be inactivated (Pinck *et al.*, 1954). This idea was supported by Craswell and Waring (1972), who found that N mineralisation increased several fold when clays were ground, the increase being greater where the clay was montmorillonitic. More recent work has suggested that it is not the texture but rather the structural relationship between particles and pores which is important in controlling nutrient cycling, turnover being slower in smaller pores (Hassink, 1992; Killham *et al.*, 1993). Ladd *et al.* (1981a) found decomposition was significantly slower in heavy clay soils than in sandier soils during the first 16 weeks after residue incorporation, and Jenkinson (1977b) observed differences throughout a 10 year period. Other workers have reported less significant, but similar, patterns (Amato *et al.*, 1987; Müller, 1988).

Soil pH

pH affects mineralisation largely through its effects on microbial activity, mineralisation being favoured in neutral to alkaline soils, but greatly decreased in acidic soils (Jenkinson, 1981). Nyborg and Hoyt (1978), using 40 soils, reported a doubling of the percentage of total soil N mineralised over a 120 day period when liming was used to increase pH from 5.0 to 6.7.

Residue Composition and Management

The decomposition of plant residues has been described approximately by two first order reactions, the first a rapid breakdown of easily decomposable components and the second a much slower decay of stabilised residues and turnover products (Amato *et al.*, 1987). The actual proportion of the total C decomposed in the initial rapid phase (about 1 year) is usually around two thirds and is remarkably similar for a wide range of crop residues (Jenkinson, 1981).

The amount of N needed by microorganisms during decomposition, and the time when the need is greatest, depends on the substrate being decomposed (Jenkinson,

1981), reflecting the variable N content of substrates and the decomposability of their components, respectively.

Much work has attempted to establish the equivalence point (Campbell, 1978), i.e. the C:N ratio at which no net mineralisation or immobilisation occurs. Harris (1988) quotes a figure of 35:1, whilst Harmsen and van Schreven (1955) report values of 20 to 25. Different researchers come up with different answers (Black, 1968), due to differences in soil and climatic conditions, and amount and composition of organic substrate (Campbell, 1978).

The equivalence point depends upon the C:N ratio of the cells being synthesised and the energy efficiency of organisms, as well as the C:N ratio of the organic residue being decomposed (Harris, 1988). Moreover, the C:N ratio is not an exact measure of available energy:N content, which is the real factor influencing the rate of decomposition (Jansson and Persson, 1982). Available energy is determined by the decomposability of the residues' components. Frankenberger and Abdelmagid (1985) observed that net mineralisation was highly significantly correlated with total N content of legumes ($r=0.93$) and C:N ratios ($r=0.88$). These authors calculated a critical N content of *ca.* 1.7% N, similar to that found by Iritani and Arnold (1960). Janzen and Kucey (1988) also considered residue N content to be the most important variable affecting decomposition, and concluded that the high correlations observed between decomposition and other plant characteristics were due to their colinearity with N content. Müller *et al.* (1988) found N release did not correlate well ($r=0.305$) with the C:N ratios of nitrogenous materials, whilst a significant negative correlation was found with lignin ($r=-0.655$). Fox *et al.* (1990) observed a better correlation ($r=-0.93$) between the plant lignin + polyphenol:N ratio and net mineralisation. Plant lignins, after incorporation into soil, degrade to polyphenols which are a main constituent of recalcitrant, N-containing humic polymers (Haynes, 1986a). Formation of these recalcitrant N compounds would reduce the rate of plant N mineralisation and thus the combination of these constituents will be the best indicator (Fox *et al.*, 1990).

Residue management, as well as chemical composition, affects mineralisation. The mineralisation of residues and manures has been shown to be largely independent of rate of addition (Jenkinson, 1977a; Azam *et al.*, 1993; Rees *et al.*, 1993). It is sometimes found that large additions of organic matter decompose more slowly than

small additions, due to N deficiency in the soil (Jenkinson, 1981) or the development of anaerobic zones (Rees *et al.*, 1993).

The rate of residue decomposition varies with depth of placement because moisture, aeration and temperature conditions change with soil depth. Generally, organic residues decompose more slowly as depth increases (Campbell, 1978) and this is supported by Richter *et al.* (1989), who found that deep ploughing of grassland slowed N mineralisation. The tool used to bury residues may also be important. If residues are buried in layers, the concentration of organic matter may lead to excessive O_2 demand and creation of anaerobic conditions, thus slowing decomposition. Swaby (1966) found that discing increased decomposition rates by distributing the residues more evenly through the soil. Lloyd (1992) found that there was no difference in mineral N release or leaching following ploughing or minimal cultivation of grass swards. Where legume residues were left on the surface (no-till), mineralisation and immobilisation rates were faster than if residues were incorporated (Varco *et al.*, 1993). Groffman *et al.* (1987) concluded that tillage practices affect the timing of N availability more than the total amount of N available, and thus play a major role in determining the fate of mineralised N.

2.2.2.2 Measurement of mineralisation

By assuming that no N is lost from the whole system and none has been added by fixation, measuring inorganic N before and after incubation in the laboratory provides a measure of net mineralisation. Stanford and Smith (1972) measured inorganic N by leaching columns with a mild extractant solution (0.01M $CaCl_2$) which has been shown to give accurate results (Keeney, 1982a). However, these authors were primarily concerned with potentially mineralisable N rather than actual rates, and field mineralisation rates are usually much lower since conditions are rarely optimal (Adams and Attiwill, 1986). Furthermore, concern has been expressed regarding the potential effects of soil alteration on mineralisation during incubations. Leaching with extractants may stimulate mineralisation due to changes in moisture status. Sieving may either increase or decrease net mineralisation depending upon soil type (Raison *et al.*, 1987), but leads to lower variability than that obtained with intact cores (Ross *et al.*, 1985). Beauchamp *et al.* (1986) found that air drying soil enhanced initial mineralisation. The different techniques used reflect the different aims of the studies (Macduff and White, 1985).

In short-term studies of agronomic crops the best measure of the N supplying capacity of soil is plant N uptake (Raison *et al.*, 1987). Because established grassland has a well developed and extensive root system, the amount of N taken up from plots receiving no fertiliser N provides a good indication of the available soil N supply (Whitehead, 1986), although N fixation by clover can lead to complications and overestimation. However, interpretation is still complex, since N supply is not the only factor affecting plant uptake (van Keulen and Stol, 1991) and some of these factors may not be controlled. If below-ground plant biomass is not harvested this can lead to significant underestimates of uptake (Saffinga, 1988). Crop sampling, unless done frequently, provides very little information about the timing of N release, and where other N loss processes are not studied simultaneously this method can only give a baseline measure of mineralisation. Fallow plots can provide more detailed information on the temporal pattern of N release but can lead to higher decomposition and mineralisation estimates than those determined with cropped soils (Jenkinson, 1977b; Janzen and Radder, 1989; Nicolardot *et al.*, 1995), and have greater potential for leaching losses. Plastic shelters have been used to eliminate leaching in some studies (Powlson, 1980; Rodgers *et al.*, 1985). Complete N balance studies (Paustian *et al.*, 1990; Vinten *et al.*, 1992) are labour intensive and require large numbers of samples to obtain reliable results.

Field incubations of soil cores provide a small-scale alternative to large-plot studies. Sequential sampling of incubated cores yields a temporal analysis of net mineralisation. The period of field exposure may be adjusted to coincide with major changes of environmental variables affecting mineralisation (Raison *et al.*, 1987). However, Adams *et al.* (1989) found that the period of containment affected estimates of mineralisation rates. The emphasis should be on adequate replication on a time scale which reflects the balance between mineralisation and immobilisation (Adams *et al.*, 1989).

Soil incubations have been carried out in plastic bags (Eno, 1960; Rees, 1989; Carsky *et al.*, 1990) and in isolated columns (Adams and Attiwill, 1986; Raison *et al.*, 1987). Such isolation can alter the soil environment, and may not be appropriate when microbial activity is high (Carsky *et al.*, 1990). Smith *et al.* (1977) suggested bags should only be used where surrounding soil moisture was relatively constant. Redman *et al.* (1989) suggested the higher moisture content in soil incubated in bags may have accounted, at least in part, for the difference between plant uptake and incubation estimates of mineralisation. Attempts have been made to equilibrate the

moisture content of incubated cores with the surrounding soil by using perforated columns (Rapp *et al.*, 1979) and this was found to limit the differential to less than 5% (Adams *et al.*, 1989).

Rees (1989) suggested that reimmobilisation of N in cores incubated in plastic bags leads to much lower estimates of mineralisation rates than from plant uptake data. Redman *et al.* (1989) did not appear to incur such problems using the same method and observed reasonable agreement with net mineralisation estimates from plant uptake plus leaching data. Rees (1989) suggested this may have been due to the lower temperatures and mineral N levels during the trial of Redman *et al.* (1989). Hatch *et al.* (1990) proposed the use of acetylene to inhibit nitrification, reducing the losses of N through denitrification and leaching.

Davidson *et al.* (1991) attempted to overcome the problem of interpreting net changes in mineral N by using ^{15}N-labelled soil cores to measure gross rates by isotope dilution. The use of ^{15}N tracer in incubated cores, rather than in larger scale microplots, cuts research costs whilst maintaining the advantages of tracer methods. Direct tracer methods determine the location of ^{15}N after a period of exposure and can be used to identify pathways of N movement (Nason and Myrold, 1991). However, this technique can involve considerable complications. In the field it is almost impossible to achieve a uniform application of ^{15}N (Appendix 6; Barraclough, 1991; Davidson *et al.*, 1991). The heterogeneity of field soils, particularly the spatial variability of mineral N, is a serious problem for the development of *in situ* applications of isotope dilution, since rate estimates are calculated using differences which amplify errors (Myrold and Tiedje, 1986).

The use of isotopically labelled plant residues provides improved sensitivity in determining the specific contribution of plant residues to the various soil N pools and has been widely used (Vallis, 1983; Azam *et al.*, 1985; Ladd and Amato, 1986; Amato *et al.*, 1987; Harris and Hesterman, 1990; Jensen, 1992). This increased sensitivity is only achieved if reliable estimates of the size of the various N pools can be obtained, a representative sample collected for analysis, and the experiment performed with adequate replication and controls (Legg and Meisinger, 1982). Fox *et al.* (1990) found that estimates of net mineralisation from legume residues using ^{15}N were significantly lower than using the difference method, and a similar pattern was reported by Azam *et al.* (1993). Fox *et al.* (1990) concluded that the difference

method was preferred for estimating mineralisation of legume N, and the ^{15}N method should be corroborated with results obtained by the difference method.

2.2.3 Nitrification

Nitrification is the biological oxidation of NH_4^+ to nitrite (NO_2^-) or NO_3^- (Haynes, 1986b). *Nitrosomonas* oxidises NH_4^+ to NO_2^- and *Nitrobacter* oxidises NO_2^- to NO_3^- (Schmidt, 1982). Populations and *in situ* activities of nitrifiers in soils are usually limited by the rate of production of NH_4^+ (Haynes, 1986b).

The lower limit for autotrophic nitrification is generally found to be around pH 4.5 (Sahrawat, 1982). In soils of pH above 7.5, toxic levels of NH_3 may result in the inhibition of the activity of *Nitrobacter* and in the accumulation of NO_2^- (Morrill and Dawson, 1967).

In general, the maximum rate of nitrification occurs at soil moisture potentials in the range of -10 (Miller and Johnson, 1964) to -33 kPa (Malhi and McGill, 1982). Below 4-5°C temperature limits nitrification (Anderson and Boswell, 1964). The optimum temperature range for nitrification in soils is usually between 25 and 35°C (Thiagalingam and Kanehiro, 1973; Myers, 1975).

Nitrification is a critical process when considering the fate of N in soils. It makes soil N available for leaching (section 2.2.5) and increases the likelihood of gaseous N losses directly (during the process itself) and indirectly (by providing the necessary substrate for denitrification). Research using ^{15}N has established that the principal source of NO_3^- in most soils is from the nitrification of NH_4^+ derived from soil organic N, not fertiliser (Addiscott and Powlson, 1992).

2.2.4 Gaseous nitrogen losses

The loss of N from soils in gaseous form may occur simultaneously due to nitrification (section 2.2.3) and denitrification. Under fertilised cut grassland, losses of N through denitrification may be greater than leaching losses, particularly in warm, wet summers (Jordan, 1989).

Biological denitrification is the dissimilatory reduction of NO_3^- or NO_2^- to gaseous forms of N by essentially anaerobic bacteria, producing molecular N_2 or oxides of N

when O_2 is limiting (Bouwman, 1990). The majority of such bacteria are heterotrophs and obtain their energy and cellular C from organic substrates (Haynes and Sherlock, 1986).

The significance of nitrous oxide (N_2O) emissions during nitrification in comparison with those from denitrification is uncertain (Haynes and Sherlock, 1986). In well aerated soils emissions of N_2O are likely to originate, at least partially, through nitrification (Freney *et al.*, 1978; Robertson, 1994).

2.2.4.1 Factors affecting gaseous nitrogen losses

Soil Aeration

Soil is heterogeneous and commonly has both anaerobic and aerobic sites (Granli and Bøckman, 1994). The aeration status of soils depends upon the balance between consumption, by microorganisms and plant roots, and supply by diffusion from the atmosphere (Smith, 1990). Denitrification may occur in anaerobic microsites within the otherwise aerobic medium in well drained soils, such as pores filled with water or sites within structural aggregates (Dowdell and Smith, 1974).

There is an inverse relationship between the rate of denitrification and O_2 concentration (Focht, 1974; Burton and Beauchamp, 1985). Nitrification is an aerobic process and as O_2 supply decreases so does the rate of nitrification (Granli and Bøckman, 1994).

Soil Moisture

The diffusion rate of O_2 through a water-filled pore is only one ten-thousandth of that through an air-filled pore. Soil water displaces air and affects levels of microbial activity and the release of available C and N substrates. Soil moisture is the most important variable controlling the potential development of anaerobic zones (Vinten and Smith, 1993).

The effect of soil moisture on gaseous N losses is shown in Figure 2.2.4.1. At low soil moisture content, emissions are low because of low microbial activity and ample O_2 supply limiting denitrification. With increasing water content, the rate of mineralisation, and subsequent nitrification, increases (section 2.2.2.1 and 2.2.3,

respectively) producing N_2O. Aerobic microbial activities are greatest at a soil water equivalent to 60% of a soil's water holding capacity (Linn and Doran, 1984; Doran *et al.*, 1990). With increasing water content, O_2 diffusion becomes increasingly impeded, increasing denitrification. As water contents increase further, NO_3^- reduction is increasingly all the way to N_2 (Granli and Bøckman, 1994) (section 2.2.4.2).

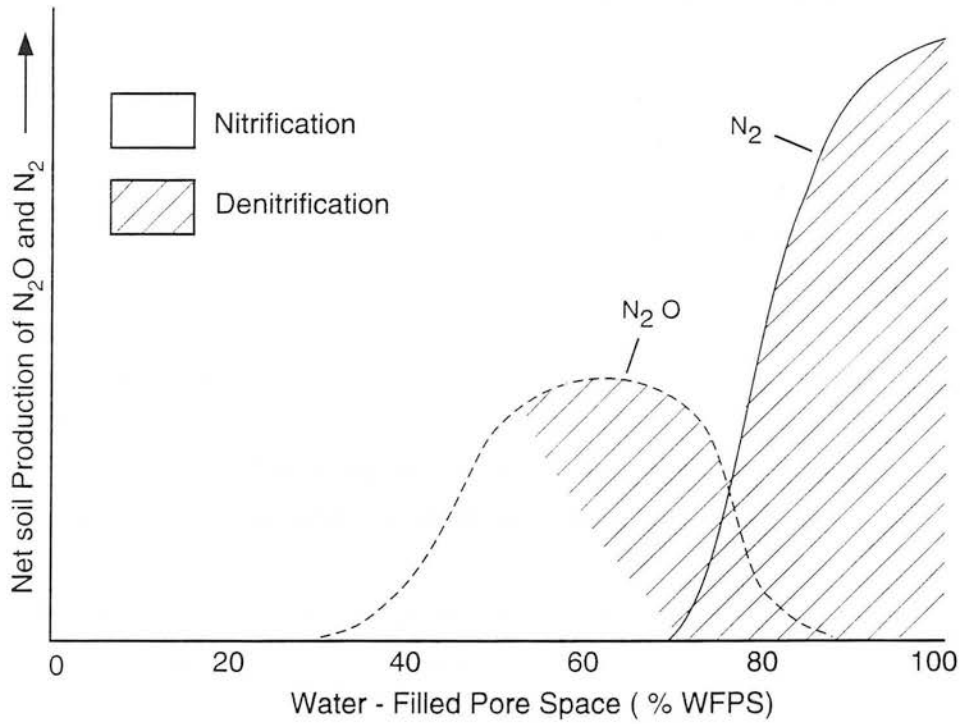


Figure 2.2.4.1 Model of the relationship between soil water-filled pore space (WFPS) and relative fluxes of N_2O and N_2. The emitted N_2O derives from both nitrification and denitrification (Davidson, 1991).

Parton *et al.* (1988) saw the production of N_2O and NO_3^- increasing simultaneously with increasing soil water at low moisture contents, indicating that nitrification was dominant; at very high soil water contents only N_2O production increased on adding water, which would suggest that denitrification is dominant.

Jordan (1989) found that large denitrification losses (>0.1 kg N ha^{-1} day^{-1}) only occurred at moisture contents greater than 25% w/w (*ca.* <30% air-filled porosity). The importance of soil water content is reflected in the numerous observations of increased gaseous N fluxes following rainfall or irrigation (Denmead *et al.*, 1979;

Smith and Tiedje, 1979; Aulakh *et al.* 1982, 1983a; Vinther, 1984; Sexstone *et al.*, 1985; Egginton and Smith, 1986; Mosier *et al.*, 1986; Jarvis *et al.*, 1991).

Soil Temperature

The denitrification rate increases with increasing temperature until an optimum is reached, above which it decreases (Aulakh *et al.*, 1992). Stanford *et al.* (1975a) found denitrification to be minimal at 0°C to 5°C, but increased ten fold between 5°C and 10°C. In the range of 10 to 35°C, a 10°C increase doubles the rate of denitrification (Dawson and Murphy, 1972; Stanford *et al.*, 1975a). The use of the Arrhenius relationship between reaction rate and temperature is not always applicable to soils, especially aerobic soils. The increase in aerobic respiration rate with temperature can result in a rapid growth in the size of anaerobic zones, while the denitrification rate per unit volume of anaerobic soil itself increases according to the Arrhenius relationship. Thus the observed rate may increase as the product of these two functions (Vinten and Smith, 1993). Jordan (1989) points out that calculated Q_{10} values may change depending on the period over which the reaction is measured, introducing additional processes competing for NO_3^-.

Stanford *et al.* (1975a) found denitrification rates changed very little between 35°C and 45°C, whilst others have found increases up to 65°C (Nõmmik, 1956; Bremner and Shaw, 1958). Goodroad and Keeney (1984) showed that N_2O production via nitrification increased as the temperature was raised from 10 to 30°C.

In the field, Denmead *et al.* (1979) found that diurnal fluctuations in N_2O emissions from an unfertilised grass sward were in phase with topsoil temperature fluctuations of 15-25°C. In contrast, Aulakh *et al.* (1983a, 1983b) found little influence of temperature in the range 10-30°C under various tillage and cropping regimes. Below 5°C only low rates of denitrification were observed under cut and grazed swards (Ryden, 1986; Jordan, 1989) or in wet soils, high in NO_3^- (Aulakh *et al.* 1983a, 1983b; Aulakh and Rennie, 1984).

Nitrogen availability

Availability of mineral N to bacteria is an important controller of the microbial processes that produce N_2O (Granli and Bøckman, 1994). Denitrification reactions are frequently reported to be first order up to 40 to 100 $\mu g\ N\ g^{-1}$ in soils (Stanford *et*

al., 1975c; Starr and Palange, 1975). Above these relatively high concentrations the rate of denitrification in soils has been shown to be independent of NO_3^- concentration (i.e. zero-order kinetics) (Blackmer and Bremner, 1978). The NO_3^--N concentration in arable topsoils is commonly between 2 and 10 μg NO_3^--N g^{-1} for most of the year (Granli and Bøckman, 1994), and is lower in unfertilised grassland (Woodmansee *et al.*, 1981).

Ryden (1986) reported that soil NO_3^--N concentrations below 5 μg NO_3^--N g^{-1} limited denitrification losses to below 0.15 kg N ha^{-1} day^{-1} on a fertilised cut ryegrass sward, whilst Jordan (1989) reported that soil NO_3^--N concentrations below 2 μg NO_3^--N g^{-1} limited fluxes to below 0.1 kg N ha^{-1} day^{-1}. In soils the diffusion of NO_3^- to the sites of denitrification can become an important limiting factor (Reddy *et al.*, 1978). Often, due to other limiting factors, NO_3^--N concentrations do not show a significant correlation with denitrification rates (Aulakh *et al.* 1983a, 1983b; Drury *et al.*, 1991).

Factors that increase nitrification rates, such as the addition of ammonium-based fertilisers (Bremner and Blackmer, 1978; McTaggart *et al.*, 1994), will tend to promote N_2O losses via this process, since the ratio of N_2O evolved to NO_3^- produced during nitrification appears to be reasonably constant (Goodroad and Keeney, 1984). Similarly, high NH_4^+ concentrations under urine patches may cause rapid nitrification (Haynes and Williams, 1992).

Soil Texture

The effect of soil texture on denitrification probably results from physical variations in soil structure, pore size, aggregation, and water infiltration rates that affect aeration, water holding capacity, and micro-environment, and may be due to other natural differences in the capacity of the soil to supply substrate (NO_3^- and C) (Aulakh *et al.*, 1992). Fine textured soils have smaller pores that more easily become anaerobic than the larger pores present in loam and sand soils (Papendick and Campbell, 1980). Nitrous oxide produced at depth can be reduced to N_2 as it moves upwards through the soil profile, and this is often the case in heavy-textured soils where diffusion is slower (Arah *et al.*, 1991).

Chaterpaul *et al.* (1980) found that denitrification rates increased from sandy loam to loam to clay loam. Webster and Dowdell (1982) found slightly higher N_2O emissions from a clay loam soil than from a better drained silt loam soil.

Soil pH

Generally, in the neutral pH range of 6 to 8 there is little effect of pH (Burford and Bremner, 1975; Stanford *et al.*, 1975c) but at soil pH values below 6 denitrification and nitrification can be strongly inhibited (Klemedtsson *et al.*, 1978; Duggin *et al.*, 1991). Weier and Gilliam (1986) found that, on average, denitrification rates from flooded soil samples increased two to three fold as pH increased with liming from 3.6-5.0 to 7.2-7.8.

Organic material

Organic C availability is one of the most important factors that affects denitrifying activity in soil (Beauchamp *et al.*, 1989), influencing microbial activity, the consumption of O_2 and the creation of anaerobic microsites. Organic compounds may be provided from native SOM, crop residues, root exudates and manures.

Burford and Bremner (1975) found that although denitrification capacity was significantly correlated with total organic C ($r=0.77$) in a range of soils, a higher correlation was obtained with water-soluble C or mineralisable C ($r=0.99$). They concluded that this reflected the importance of the supply of available C, as suggested in earlier work (Bremner and Shaw, 1958).

Drying and wetting cycles have been shown to increase denitrification capacity (Patten *et al.*, 1980) by increasing the levels of available C (Rolston and Liss, 1989). Bijay-Singh *et al.* (1988) found that while air-dried soils showed the highest correlations between denitrification potential (DNP) and water-soluble C and aerobically mineralisable C, field-moist soils showed the closest correlation of DNP with C mineralised under anaerobic conditions. The authors suggested that, in the field, denitrification may often be limited by the amount of C susceptible to mineralisation under anaerobic conditions.

Webster and Goulding (1989) found that autumn denitrification rates were markedly higher where soil C content was higher due to long-term manure application than where nutrients had been supplied only as inorganic fertilisers.

In the shorter term, addition of animal wastes may promote denitrification due to decreased O_2 supply and increased microbial activity. Manure application increases the soluble C content of soil (Meek *et al.*, 1974; Rolston and Liss, 1989). Guenzi *et al.* (1978) concluded that, in cattle manure amended soils, biological O_2 demand was sufficient to cause anaerobic microsite development.

Fresh plant residues can provide the C necessary for a strong respiratory sink for O_2, can become a source of NO_3^- through mineralisation of organic N compounds present in the residue, and provide reductant for the denitrifier (Groffman *et al.*, 1988). Parkin (1987) found that a fragment of organic material (80 mg) accounted for 85% of denitrification activity in a 98 g soil core, and concluded that where O_2 consumption was high enough anaerobic "hotspots" may occur regardless of diffusional constraints caused by aggregates.

Bremner and Blackmer (1981) reported that the N_2O emission rate increased as the C:N ratio of added organic amendments decreased. Aulakh *et al.* (1991b) found the same inverse relationship between C:N ratios (from 8 to 82) and denitrification when adding various crop residues, particularly during the first 8-10 days following incorporation. When crop residues low in N are incorporated in conjunction with mineral fertiliser, denitrification losses have been found to increase (Aulakh *et al.*, 1984).

Plants

Root exudates and sloughed off root cells promote large populations of denitrifiers (Woldendorp, 1963). In C-limited systems, increased microbial respiration and root respiration will tend to deplete soil O_2 and in these circumstances the presence of plants tends to increase the rate of denitrification of added NO_3^- (Stefanson, 1972). However, under field conditions, growing plants generally compete with denitrifiers for NO_3^- and water, reducing the rates of denitrification (Aulakh *et al.*, 1982, 1983b). Terry *et al.* (1981) reported greater emissions from fallow than cropped soils, and Aulakh (1983a) found denitrification losses were 2 to 7 times greater in fallow than wheat treatments.

Smith and Tiedje (1979) confirmed that when soil NO_3^- concentrations are high denitrification rates are increased in the rhizosphere, whereas when NO_3^- concentrations are low denitrification rates are decreased in the presence of roots. In grassland, where levels of mobile C are considered to be high (Jarvis *et al.*, 1991), the uptake of NO_3^- by roots is probably the more important factor.
Where plants are cut or damaged and roots remain in the soil N_2O emissions may increase, due to the release of readily available organic matter from roots (Beck and Christensen, 1987).

2.2.4.2 Factors affecting the N_2:N_2O ratio

A major factor contributing to the variability of N_2O emissions due to denitrification is the wide variation in the fraction of the total gaseous flux emerging as N_2O (Vinten and Smith, 1993).

Firestone *et al.* (1979) found that the N_2:N_2O ratio increased with decreasing O_2 availability, and the overall rate of gas production increased. The same pattern has been observed in poorly structured soils, or at high soil water contents (Focht, 1978; Smith, 1990; Weier *et al.*, 1993). The main reason is that decreased O_2 diffusion increases anaerobic zones, whilst decreased N_2O diffusion retards its escape from the soil, increasing the likelihood of further reduction (Arah and Smith, 1990).

Increasing temperature tends to increase the N_2:N_2O ratio (Nõmmik, 1956; Keeney *et al.*, 1979), although Bailey and Beauchamp (1973) found little effect. Several workers have found that the N_2:N_2O ratio decreases with increasing NO_3^- (Blackmer and Bremner, 1978; Firestone *et al.*, 1979; Vinther, 1984; Weier *et al.*, 1993). Increasing C availability has been reported to lead to a more complete reduction of NO_3^-, increasing the N_2:N_2O ratio (Smith and Tiedje, 1979; Weier *et al.*, 1993).

The reduction of N_2O is much more sensitive to acidic conditions than is that of NO_3^- (Granli and Bøckman, 1994). Consequently the N_2:N_2O ratio strongly decreases when pH falls below pH 5 (Nõmmik, 1956; Blackmer and Bremner, 1978).

2.2.5 Leaching

Leaching is defined as the transport of N in water-soluble forms out of a defined soil volume, usually the root zone, into a subsoil region (White, 1988). Leaching is often the most important channel of N loss from cultivated field soils other than that accounted for in crop uptake (Legg and Meisinger, 1982).

Organic N has a generally low mobility in soils. Ammonium is unlikely to be leached because NH_4^+ ions are held in the soil by the processes of cation exchange and fixation within clay lattices. In contrast to NH_4^+, the NO_3^- anion is not adsorbed by the negatively charged soil colloids and is thus susceptible to diffusion and mass transport with soil water (Cameron and Haynes, 1986). However, this electrostatic repulsion means that a proportion of the soil water does not participate in NO_3^- leaching. The effective pore volume is often up to 10-20% less than the water content of the soil (Wild, 1981). Anion exclusion from very fine pores may cause a NO_3^- pulse to move slightly faster than the accompanying water (Cameron and Wild, 1982), but the effect is usually insignificant compared with the bypassing effect created by soil structure (Dyson and White, 1987) (section 2.2.5.1).

2.2.5.1 Factors affecting leaching losses

Season and Weather

Weather is a major control of NO_3^- leaching losses from the soil system. It directly affects leaching by determining the balance between evapotranspiration and precipitation, and hence the net downward flux of water. Indirectly, weather affects the pool of available soil NO_3^- via its role in mineralisation (section 2.2.2.1) and other processes competing for NO_3^- (section 2.2.4.1) (White, 1988).

In most of lowground UK, there is usually a period during the summer when evapotranspiration exceeds rainfall and therefore leaching is minimal. However, intense heavy rainfalls may cause leaching by macropore flow, the amount of loss being dependent on the location of soil NO_3^-. As precipitation increases and evapotranspiration decreases soil moisture levels gradually reach field capacity during the autumn (White, 1988). After this, winter rainfall readily leaches any NO_3^- present in the soil profile, since there is a large excess of rainfall over evapotranspiration and

a low rate of N uptake by crops (Shaw, 1962; Kilmer *et al.*, 1974). Spring rainfall determines whether newly mineralised NO_3^- is quickly leached (Williams, 1975).

Nitrate concentrations in the drainage from structured clay soils are commonly highest in the first flows in autumn and early winter and subsequently decline (Harris *et al.*, 1984; Haigh and White, 1986). This pattern is often most marked if the previous summer has been unusually hot and dry (Garwood and Tyson, 1977; Foster and Walling, 1978). In contrast, many workers have observed a close positive relationship between percolate volumes and N concentrations in percolate (Tomlinson, 1971; Chichester, 1977). Large protracted flows of low concentrations of NO_3^- can be considerably more significant to total loss than short periods with high NO_3^- concentrations.

Soil Texture and Structure

The field capacity or volumetric water content above which water moves freely under the influence of gravity is generally greater in clay soils than sandy soils (White, 1988). For a given rate of water application water will therefore percolate at a higher mean velocity through a sandy soil than a clay, displacing NO_3^- to a greater depth (Kolenbrander, 1969; Cooke, 1976). Nitrate losses are normally greater from poorly structured sandy soils than from coarsely structured clay soils (Avnimelech and Raveh, 1976; Sommerfeldt *et al.*, 1982). Webster *et al.* (1986) found that sandy loam soils leached more N, 74 (range 34-102) kg N ha^{-1}, than clay soils, 41 (range 15-73) kg N ha^{-1}, over 6 years.

However, it is over-simplistic to assume that NO_3^- always leaches more rapidly from sandy than clay soils, since the effect of texture is modified by structure and the microscale distribution of NO_3^- in the soil (White, 1985). Variations in pore size, in the spatial distribution of pores and in their continuity all contribute to irregular movement of water down the soil profile (Tyler and Thomas, 1981). Earthworm activity, root growth, freezing and thawing, and wetting and drying cycles can lead to the development of a network of connected pores (Cameron and Haynes, 1986). Recognition of this hydrodynamic dispersion, particularly in structured soils, has led to the concept of mobile and immobile water. The immobile water is that retained in peds, from which NO_3^- can only be transferred to the mobile phase by diffusion across the mobile-immobile water interface (Vinten and Smith, 1993). In a coarsely structured soil receiving a rapid application of water, over half the water may move

through the macropore system (Quisenberry and Phillips, 1976; Thomas *et al.*, 1978). When infiltrating water contains a high NO_3^- concentration, then macropore flow will lead to extensive leaching at a faster rate than predicted by the convective-diffusive-dispersive mechanism (Addiscott and Cox, 1976; Barraclough *et al.*, 1983). Where soil NO_3^- is predominantly the result of mineralisation and nitrification within aggregates, or where NO_3^- has diffused into aggregates, then water moving in macropores will carry little NO_3^- with it (Wild, 1972). Nitrate within aggregates is relatively protected during individual events, but between events it diffuses to the aggregate exteriors where it can be leached (White, 1988).

Land Management

Large amounts of N are cycled in mature, undisturbed natural ecosystems and both inputs and losses are small (Cameron and Haynes, 1986). Agricultural ecosystems tend to be continually disturbed and, given that they are managed more intensively, leaching losses can be large (Wild and Cameron, 1980).

Lack of vegetation for at least part of the year is a key factor stimulating NO_3^- leaching from cropping systems since, where conditions are favourable for rapid mineralisation and nitrification, high levels of NO_3^- will accumulate in the surface of soils (Cameron and Haynes, 1986). Vegetation reduces the propensity for NO_3^- to be leached, but annual crops are generally less effective than perennials such as grasses (Kolenbrander, 1981; White, 1988). Rooting habits of plants can exert a great influence on NO_3^- leaching through the root zone, since plants remove both NO_3^- and water from the soil (Singh and Sekhon, 1979). Bergström (1987) reported higher leaching losses from barley than from leys, due to greater drainage and N available for leaching following barley. Widdowson *et al.* (1987) showed that the amounts of mineral N following crop harvest can vary greatly between soils, even after the same crop.

Where grassland is grazed the situation is significantly altered. Ryden *et al.* (1984) reported that leaching losses were 5.6 times higher under a grazing than a cutting regime. Haigh and White (1986) reported that under grazed grassland receiving more than 100 kg N ha^{-1} yr^{-1}, 40% of the discharge from tile drains exceeded the EU limit for NO_3^- concentrations. This is due to the low retention of ingested N by grazing animals (Whitehead, 1970a) and the spatially concentrated return of this N in amounts larger than grass uptake capacity (Ball and Ryden, 1984), particularly late in

the growing season (Thomas *et al.*, 1988; Cuttle and Bourne, 1992, 1993). Recent studies (Titchen *et al.*, 1989; Lord, 1992) have assessed the potential of earlier animal removal to reduce leaching losses from grazed swards.

Garwood and Ryden (1986) reported that leaching losses from grazed grass-clover swards were only one seventh of those from fertilised grass swards. Owens *et al.* (1994) reported a *ca.* 50% decrease in subsurface flow NO_3^- concentrations within 3 years following a change in N supply from fertiliser to legume N, although herd size was reduced by 29%. Other workers have found losses from high clover content swards to be equivalent to those from heavily fertilised grass (Macduff *et al.*, 1990). Parsons *et al.* (1991) suggested that moderate clover contents (10-20%) could maintain acceptable levels of production whilst minimising environmental impacts. Cuttle *et al.* (1992) found that leaching losses decreased as clover content and, critically, stocking rates decreased.

The ploughing out of grasslands to produce arable land can lead to high concentrations of NO_3^- in the soil profile which may subsequently be leached (Low *et al.*, 1963; Cameron and Wild, 1984). The cultivation of permanent pasture since the 1939-1945 war has been linked to subsequent elevated concentrations of NO_3^- in water (Foster *et al.*, 1982). The later the ploughing is carried out in the autumn, the less N is at risk from leaching (Francis *et al.*, 1992; Lindén and Wallgren, 1992). Stopes and Philipps (1992) reported that whilst spring ploughing of leys led to lower leaching losses than ploughing the previous autumn, any benefits were eliminated the following winter when the pattern of leaching losses was reversed.

2.2.5.2 Measurement of leaching

Techniques for effectively monitoring the leaching of pollutants through the unsaturated zone of soils are needed to provide an early warning of potential groundwater contamination (Barbee and Brown, 1986). To estimate the amount of NO_3^- leaching, two variables, the quantity of water participating in leaching and the flux-averaged NO_3^- concentration in that water, must be known (Magesan *et al.*, 1994).

Lysimeters can be used to study the quantity and quality of water moving downward through a soil profile. Monolith lysimeters, the most commonly used (Goulding and Webster, 1992), are large undisturbed soil blocks cut from the ground and encased to

allow all drainage to be collected and analysed (Belford, 1979). For optimal performance, lysimeters should be deep enough to represent the whole of the active soil profile including vegetation rooting depth, and have a large enough horizontal cross section to contain a representative soil unit (Thorburn, 1992). The use of detachable upper collars allows normal field operations (Lord, 1990). However, lysimeter conditions are not entirely natural because the soil-air interface at the lysimeters' base prevents suction from the subsoil (Wagner, 1962; Webster *et al.*, 1993). The use of tension lysimeters overcomes this problem by creating suction equivalent to a soil column, and was found especially useful in coarse soils (Wallihan, 1940; Colman, 1946) although Webster *et al.* (1993) suggest suction is difficult to estimate and apply. Whilst a 'settling in' period is often required (Bergström, 1987; Goulding and Webster, 1992; Thorburn, 1992; Webster *et al.*, 1993), lysimeters are considered to be the standard against which other methods are tested (Goulding and Webster, 1992) and are the definitive technique employed in measuring the leaching propensity of pesticides for registration purposes (Thorburn, 1992).

Field drainage installations offer the best technique for measuring leaching where deep seepage is minimal and they are invaluable for estimating leaching from structured clay soils (Goulding and Webster, 1992). They can overcome some of the problems of spatial variability, lateral flow and soil disturbance associated with other methods (Goulding and Webster, 1992) and they allow research in conditions in which normal agronomic operations can occur (Vinten and Smith, 1993). However, partial recovery of drainage and lateral movement of water below the depth of hydrological isolation can cause problems (Bergström, 1987; Vinten and Smith, 1993).

Such methods are expensive; the use of ceramic cups in conjunction with neutron probe moisture measurements, or other drainage estimation techniques is cheaper, and yields similar information to lysimeters under more natural soil moisture conditions (Wagner, 1962). Porous cup samplers are widely used for measuring NO_3^- leaching from agricultural land (Lord and Shepherd, 1993).

Porous cups are relatively simple to install, involve negligible disturbance of the soil profile and allow continuous sampling, at different depths within the same profile (Grossman and Udluft, 1991). However, despite the fact that they have been used for many years (Briggs and McCall, 1904), the installation techniques, suctions applied and duration of sampling periods are far from uniform (Wagner, 1962; Harris and

Hansen, 1975; Alberts *et al.*, 1977; Talsma *et al.*, 1979; Cuttle *et al.*, 1992). The solute concentrations detected in cup samples are not significantly affected by the applied suction (Hansen and Harris, 1975; Beier and Hansen, 1992; Lord and Shepherd, 1993) or the angle of suction probe installation (Lord and Shepherd, 1993).

Porous cup samplers are limited because of the inability to define *a priori* the sampling rate, sampling zone (Morrison and Lowery, 1990) and soil water fractions from which water is taken. If the sampling rate and sampling zone values were available for a given sampler for a range of soil hydraulic conditions, the optimal sampler location could be determined (Morrison and Lowery, 1990). Wagner (1962) found that sample volumes were dependent not only on the water content of the layer sampled but also on those of adjacent layers, presumably due to considerable capillary forces. Assuming a constant suction, the radius of influence of a porous cup may be several metres (van der Ploeg and Beese, 1977). Narahsimhan and Dreiss (1986) modelled a more realistic falling suction and stressed the importance of the sampling probes' volume in affecting the cups' radius of influence.

The potential gradient generated by a suction cup acts on all pores. However, the extent of water movement and flow velocities are highly dependent on the diameter of the pores (Grossman and Udluft, 1991). Under stationary conditions in homogenous sediments there is no reason to assume that suction cups should extract water only from pores of a certain size (Grossman and Udluft, 1991). Soils are not homogenous and therefore the proportion of the sample from large pores (not macropores) may be too high, particularly at high suctions (Severson and Grigal, 1976). van der Ploeg and Beese (1977) suggested the use of larger samplers using a low applied suction, but even at a small suction seepage rates are faster than those which would occur under free drainage (Barbee and Brown, 1986). A representative sample would have to have a distribution of the sample volume collected through time identical to the soil drainage rate curve. Any difference between the curves will lead to bias (Hansen and Harris, 1975).

Porous cups have been shown to be a reliable tool for measuring NO_3^- leaching on sandy (<10% clay) soils (Lord and Shepherd, 1993). Such soils have a small range of mostly large pores in which differing solute concentrations are unlikely to develop (Webster *et al.*, 1993). Djurhuus (1990), in an intensively sampled study, found no significant difference between soil water NO_3^- concentrations measured by soil

coring and porous cups. Webster *et al.* (1992, 1993) also found no significant difference between the amounts of leaching determined using coring and cups respectively, although NO_3^- concentrations were often smaller in soil cores. Several workers have found good agreement with solute concentrations in lysimeter drainage and porous cup samples (Hansen, 1991; Jacobsen *et al.*, 1992; Thorburn, 1992; Webster *et al.*, 1992, 1993).

Porous cups may inadequately represent the field soil solution if inter-ped pores channelling solutions at high potentials bypass porous cups completely (Shaffer *et al.*, 1979; Barbee and Brown, 1986). In well structured soils such bypass flow is far more likely (Shaffer *et al.*, 1979; Tyler and Thomas, 1981; Barbee and Brown, 1986; Barraclough *et al.*, 1992). Workers have used free drainage samplers (Barbee and Brown, 1986), zero tension lysimeters (Haines *et al.*, 1982; Russell and Ewel, 1985; Hendershot and Courchesne, 1991) and sampling of lateral flow (Joslin *et al.*, 1987) in an attempt to overcome this problem.

Field soil heterogeneity of hydraulic properties, mineral and organic compound distribution and floral and faunal activity (Grossman and Udluft, 1991), lead to highly variable solute concentrations, both spatially and temporally (van de Pol *et al.*, 1977; Amoozegar-Fard *et al.*, 1982). Unless a constant suction is applied, porous cups may fail to collect leachate at critical times (Barbee and Brown, 1986). Hansen and Harris (1975) suggest that variability of ±30% should be expected in field studies when sampling concentrations around 4.5 $\mu g\ NO_3^-$-N ml^{-1}, and could be reduced by stratifying groups according to intake rate and collecting samples over a uniform sampling period, as short as possible, to reduce sampler variability. This does not, however, imply greater representativeness. Alberts *et al.* (1977) calculated that 10 replicates would be needed to get an estimate of NO_3^- concentrations within 30% of the true mean, whilst van de Pol *et al.* (1977) calculated that 24 probes were required to estimate pore water velocities within 25% of the true mean, even with a constant infiltration rate. Lord and Shepherd (1993) suggested that 20-25 probes would be required to detect differences of 25% in treatment means, although more would be required on heavily manured plots. Reliable quantification requires knowledge of field variability (Biggar, 1978), frequency distributions (van de Pol *et al.*, 1977) and consideration of the chemical being sampled (Beier and Hansen, 1992). Cuttle (1992) and Cuttle *et al.* (1992) emphasised the importance of considering the source of NO_3^- leaching, reporting highly skewed distributions under grazed pastures as well as additional complications caused by livestock "camping" areas.

Soil coring is the simplest and cheapest (per single measurement) method to estimate leaching losses (Goulding and Webster, 1992), and is widely used (Mohammed *et al.*, 1984; Neeteson *et al.*, 1989; Barraclough *et al.*, 1992; Lloyd, 1992). However, the method is imprecise because it requires assumptions regarding mineralisation, nitrification, denitrification and immobilisation unless measurements are very frequent (Lord and Shepherd, 1993). Furthermore, because soil sampling is site-destructive, time-dependent changes may be completely masked by spatial variability in NO_3^- concentrations (Lord and Shepherd, 1993). Soil coring is generally found to have higher coefficients of variation, perhaps due to the smaller volume of soil sampled by coring, and therefore for a given level of precision more cores are needed than cups (Djurhuus, 1990; Lord and Shepherd, 1993). The technique is of most value to complement and clarify other measurement techniques (Goulding and Webster, 1992).

Both porous cups and soil coring require an accurate estimate of drainage in order to be applied to the calculation of leaching loads. The identification of the onset of drainage is critical (Webster *et al.*, 1993), especially when initial concentrations are high (Lord and Shepherd, 1993). Effective operation of porous cup samplers at low suctions may be taken as an indication of an approach to field capacity. Alternatively, soil moisture can be measured using soil sampling, neutron probes, time domain reflectometry or soil water modelling (Thorburn, 1992). Webster *et al.* (1992) found good agreement between overwinter lysimeter drainage and that calculated using site-specific meteorological data, whereas MORECS data taken over a 20 km grid was too imprecise. These authors suggested that differences in lysimeter drainage due to lack of subsoil matric suction should be accounted for if lysimeters were used as an independent measure of drainage.

Whilst not being a direct measure of NO_3^- leaching, ^{15}N-labelling is invaluable because it allows detection of the source of N lost from farming systems (Powlson and Barraclough, 1993) either from fertiliser (Dowdell and Webster, 1980; Macdonald *et al.*, 1989) or from residues (Ladd *et al.*, 1981b; Müller, 1987; Jensen, 1992).

2.3 CONSEQUENCES OF LOSSES OF NITROGEN

The leaching of NO_3^- in drainage water from agricultural soils is undesirable as it represents an economic cost to the grower as well as posing a risk of pollution to water resources (Thorburn, 1992).

The loss of NO_3^- into water represents a potential health risk. The incidence of methaemoglobinaemia in infants (a condition in which the capacity of the blood to carry O_2 is reduced) may increase as NO_3^- concentrations in water increase (NRC, 1978). Only 14 cases have been reported in the last 42 years which were attributable to NO_3^- in drinking water. The last reported confirmed case in the UK was in 1972 (Brown, 1992). There is also circumstantial evidence relating to NO_3^- exposure and the incidence of cancer (NRC, 1978).

Agriculture is the main source of NO_3^- pollution of water in the EU, normally accounting for over 60% of total NO_3^- loss to water (Tunney, 1992). Nitrate concentrations in all public water supplies must be kept below the maximum admissible concentration permitted by the EU of 11.3 μg NO_3^--N ml^{-1}. The scientific basis of this limit with respect to human health has been extensively debated (House of Lords, 1989) and is clearly deficient (Garrett *et al.*, 1992). The Department of the Environment concluded in 1988 that land use control, rather than water treatment, would prove the cheaper option (DoE, 1988). MAFF has set up Nitrate Sensitive Areas in some catchments where farmers are encouraged to adopt practices to minimise N losses from their soil (Archer, 1992; Webster *et al.*, 1993).

In some regions of the UK, mostly in southern and eastern England, water supply companies are having difficulty meeting the EU limit (Croll, 1990) and, in 1983-84, it was exceeded in 125 groundwater sources supplying 1.8 million people (Lean, 1990). In 1980 and 1960, 90 and 60 groundwater supplies, respectively, exceeded the EU limit (DoE, 1986).

In addition to potential human health risks, increased supply of N to surface waters, particularly lakes and estuaries, can increase biological activity to eutrophic levels. Phosphorus is more frequently the limiting nutrient involved in eutrophication in fresh waters, thus the role of N can seldom be quantified (Keeney, 1982b). In marine systems NO_3^- is usually the limiting nutrient and there is evidence of increased coastal marine water eutrophication due to riverine NO_3^- inputs (Owens, 1993).

Soil microbial processes are the principal sources of N_2O and, whilst estimates are highly uncertain, account for about 65% of the total source (IPCC, 1995). As the lifetime of N_2O in the atmosphere is about 150 years, changes in its concentration will have a long-term effect (Bouwman, 1990). The concentration in the atmosphere is currently increasing at about 0.2-0.3% $year^{-1}$ (Granli and Bøckman, 1994). This increase causes concern for two reasons. Nitrous oxide is a greenhouse gas, contributing about 5% of the anthropogenic greenhouse effect (Bouwman, 1990) and is the major contributor to stratospheric O_3 depletion (Prinn, 1994) which could lead to increased exposure of the biosphere to UV radiation (Crutzen, 1974).

3 SITES, MATERIALS AND METHODS

3.1 FIELD TRIALS

3.1.1 Beechgrove

3.1.1.1 History

The Beechgrove trial site was located on the Bush Estate, Midlothian, about 15 km south of Edinburgh (NT243628), altitude 185 m. The land capability class for agriculture is 3.1, with wetness limitations (Bibby *et al.*, 1982). The trial was established in May 1987 to assess the potential of grass-clover swards for lamb production, compared to fertilised grass swards (Swift *et al.*, 1993). Swards were continuously grazed from April to September, using practices which encouraged the spread of clover and enhanced clovers contribution towards herbage production (Swift and Vipond, 1991).

The experimental paddocks included swards of perennial ryegrass (*Lolium perenne* L. cv. Contender) and, perennial ryegrass (*Lolium perenne* L. cv. Condesa) (37 kg seed ha^{-1}) sown with small-leaved white clover *(Trifolium repens* L. cv. Aberystwyth S184) (3 kg seed ha^{-1}). Adjacent holding paddocks of the same sward mixes were also established. The grass paddocks received 150 kg N ha^{-1} as four split applications (60 March, 30 May, 30 June, 30 July) in 1991, and 90 kg N ha^{-1} in 1992 (60 March, 30 May) prior to being fenced off from grazing animals. The grass-clover paddock received no fertiliser N but was supplied with supplementary P and K to meet clover requirements.

Swift *et al.* (1983) examined grassland on 'stock rearing with arable' farms in eastern Scotland and found that two-thirds of the grass was rotational, under 7 years old. The ages of the Beechgrove swards at ploughing were therefore representative of a considerable proportion of farm systems in the region. Since white clover, grown in mixed swards, usually with *Lolium perenne*, is the principal forage legume in the UK (Hopkins *et al.*, 1994), and L cv. Aberystwyth S184 is highly suited to sheep grazed pastures, the Beechgrove swards were highly representative of grassland used for sheep production.

3.1.1.2 Soils

The soil was of the Winton series and classified as an imperfectly drained brown earth, which has a clay loam topsoil over a silty clay loam to clay loam subsoil (Ragg and Futty, 1967; Vinten *et al.*, 1991). A full profile description is given in Table 3.1.2.2. Supplementary details of the soil profile were gained from deep cores taken throughout the experimental period (section 3.6.3). In most cores the A/B horizon boundary occurred at about 35 cm.

The Winton series and Macmerry series (section 3.1.3) are the most important series in the Winton Association. The Winton Association is considered to be included within the Rowanhill Association which is widespread within the Midland Valley of Scotland (Ragg and Futty, 1967). The soil conditions experienced during the field trial at Beechgrove, and the other trial sites, are therefore representative of a large area of Central Scotland.

3.1.1.3 Treatments and hypotheses

The key objectives of the Beechgrove trial were to:
a) Quantify the amount and timing of the release of mineral N from ploughed out grass and clover-rich swards during the first two years after cultivation.
b) Study the fate of this mineral N during the first two years after cultivation, quantifying:

i) potential plant uptake;
ii) gaseous losses of N;
iii) leaching losses of N.

c) Assess the impact of sward composition and sward management on the soil N cycle following incorporation of grassland.

These objectives were met through the establishment of separate blocks, isolated in the grass and the grass-clover holding paddocks. The following hypotheses were tested:

a) Nitrogen mineralisation following incorporation of grass-clover swards is greater than that from grass swards.
b) Nitrogen mineralisation following incorporation of previously grazed swards is greater than that from previously cut swards, regardless of sward composition.

c) Undisturbed, cut grassland is subject to smaller losses of N (via leaching and/or gaseous losses) than disturbed grassland, regardless of sward composition.
d) Following incorporation, losses of N from the soil system (via leaching and/or gaseous losses) from grass-clover swards are greater than from grass swards.

Table 3.1.1.3 lists the treatments that were applied, full details of which are given in section 3.1.1.4. The treatments were replicated three times and laid out in a Youden Square design in each block (Cochran and Cox, 1957). The plots were 5 x 3.75 m and 5 x 4.75 m in the grass and grass-clover blocks, respectively. Treatments are referred to in the subsequent text using both abbreviations and descriptions. In referring to similar treatments in the two blocks a general descriptive term is used. For example, PGC92 and PG92 are referred to as the resown 1992 treatments.

Table 3.1.1.3 Treatments applied at the Beechgrove field trial, 1992-3.

Treatment description	Abbreviation
Grass-clover block	
Grass-clover sward ploughed out 1992, resown to ryegrass	PGC92
Grass-clover sward ploughed out 1992, left fallow	PGCF
Continued undisturbed grass-clover sward	CGC
Grass-clover sward ploughed out 1993, resown to ryegrass 1993	PGC93
Grass-clover sward ploughed out 1993, left fallow	EP[a]
Grass block	
Grass sward ploughed out 1992, resown to ryegrass	PG92
Grass sward ploughed out 1992, left fallow	PGF
Continued undisturbed grass sward	CG
Grass sward ploughed out 1993, resown to ryegrass 1993	PG93

3.1.1.4 Agricultural operations

On 10 June 1992 both swards were cut to a height of about 4 cm and vegetation cuttings removed. The resown 1992 and fallow treatments (PGC92, PGCF, PG92 and PGF) were ploughed out on 10 June to a depth of about 23 cm, and rotavated on 22 June. Prior to 22 July 1992, the trial area was not fenced off from sheep in the surrounding paddocks. After a period of heavy rain sheep trampling compacted the grass-clover plots, and so all previously rotavated plots were rerotavated on 20 July. The resown 1992 treatments (PGC92 and PG92) were then ridge rolled, hand sown

with ryegrass (*Lolium perenne* L. cv. Parcour), raked over and flat rolled on 20 July. The fallow treatments (PGCF and PGF) received exactly the same operations except no ryegrass seed was sown.

The fallow treatments were not immediately sprayed with herbicide to control any regrowth, and considerable weed growth occurred. These treatments were sprayed with Glyphosate on 23 December 1992 and again on 18 June 1993.

Herbage on all treatments, except the fallow treatments, was cut using a rotary mower and all cut herbage was removed from the plots. Throughout the trial cuts took place on the same dates, and to the same height, as hand-cut vegetation sampling (section 3.5.2.1).

In 1993, the resown 1993 treatments (PGC93 and PG93) were ploughed out on 11 May, one month earlier than in 1992, to a depth of about 25 cm. The plots had been cut to a height of about 4 cm on 4 May. Plots were rotavated on 26 May and raked the following day. Plots were hand sown with Parcour ryegrass (36 kg seed ha^{-1}), raked over and flat rolled on 7 June.

An attempt to plough the PGC93 treatment was made on 5 April but was aborted due to excessive tractor wheel slippage. This resulted in the creation of the EP plot (Table 3.1.1.3) and plot PGC93 2 being moved to a new position. The EP plot was rotavated on the same date as the resown 1993 treatments.

3.1.2 Glencorse

3.1.2.1 History

The Glencorse trial site was situated at Glencorse Mains (NT236628), about 15 km south of Edinburgh at an altitude of 207 m. The land capability class for agriculture is 3.2, with wetness limitations (Bibby *et al.*, 1982). Eight hydrologically isolated plots, 15 x 20 m, were installed in 1987 (Vinten *et al.*, 1991). Prior to 1989 cropping was mainly winter and spring barley. More recent cropping history is shown in Table 3.1.2.1. The previous fertiliser application rates may have added to variability between plots in treatments applied in 1993-4, most notably in the AN treatment (Table 3.1.2.4).

3.1.2.2 Soils

Glencorse is an imperfectly drained clay loam of the Winton Series (Ragg and Futty, 1967; Vinten *et al.*, 1991). Details of the soil profile are given in Table 3.1.2.2. The colours of the soils of this series are highly variable and soil hues change quite markedly over a few metres (Ragg and Futty, 1967).

Mottling is intense in the B horizons. The dry bulk densities used to convert soil mineral N analyses were 1.24, 1.4 and 1.5 g cm^{-3} for the 0-20, 20-40 and below 40 cm depths, respectively.

Table 3.1.2.1 Previous cropping, treatments and N fertilisation at the Glencorse field trial site, 1989-92.

Year	1989	1990	1991	1992
Crop	Spring Barley	Spring Barley	Spring Barley / Grass[c] / Grass-clover[c]	Spring Barley / Grass / Grass-clover
Sowing	20 April	13 April	12 April	28 March
Harvest	22 August	4 September	3 October	25 September
Ploughing	September[a]	September[b]	-	January
N Split	Sowing	1:1, sowing / tillering	1:1, sowing / 30 May	Tillering 5/5
		Fertiliser N application (kg N ha^{-1})		
Plot 1	120	150	0	0
Plot 2	120	0	0	0
Plot 3	120	180	180	180
Plot 4	0	120	0	0
Plot 5	120	90	0	0
Plot 6	120	150	150	150
Plot 7	0	210	210	210
Plot 8	120	150	0	0

[a] Plots 2-7 were chisel ploughed and subsoiled. Plots 1 and 8 were ploughed in February 1990.
[b] Chisel ploughed.
[c] Sown in September. Previously under stubble.

3.1.2.3 Agricultural operations

Plant tops on the grass and clover-rich swards were cut on 6 July 1993 and chopped on 14 July. Treatments PC and PG (Table 3.1.2.4) were ploughed out on 14 July and rotavated on 21 July. Treatments CN and AN were rotavated on 26 July. On 10 August all treatments were hand sown with 20 kg seed ha^{-1} (*Lolium perenne* L. cv. Contender) and flat rolled. Treatments CN, AN and PG were fertilised by hand on 12 August.

Table 3.1.2.2 Typical profile description of soil at Glencorse (from Vinten *et al.*, 1991).

Horizon	Depth (cm)	
Ap (g)	0-30	Dark greyish brown (10YR 4/2) with few fine medium yellowish brown (10YR 5/6) mottles; clay loam; moderate medium subangular blocky; firm; moist; low organic matter; common fine roots; common small subangular stones; clear change to:
B (g)	30-72	Brown (7.5YR 5/2) with common medium and fine yellowish red (5YR 5/8) mottles and few fine grey (10YR 6/1) gley streaks; silty clay loam to clay loam; weak medium angular blocky; firm; moist; no organic matter; very few fine roots; few small subangular stones; gradual change to:
B_2 (g)	72-82	Dark brown-brown (7.5YR 4/4) with common fine strong brown (7.5YR 5/8) mottles and common to many medium and coarse grey (N/6) gley patches and streaks; silty clay loam; weak medium angular blocky to massive; firm; moist; no organic matter; no roots; few small subangular stones; clear change to:
BC (g)	82-110+	Reddish brown (5YR 4/4) with common fine strong brown (7.5YR 5/8) mottles and many coarse grey (N/6) gley patches; common small blocky manganiferous concretions; sandy clay loam; weak medium subangular blocky to massive; firm; moist, but becoming wet; no organic matter; no roots; common medium stones; few large stones; mainly sandstone with some igneous.

3.1.2.4 Treatments and hypotheses

Previous sward management on the Glencorse trial (section 3.1.2.1) was more representative of a set-aside system than a productive grassland system such as that at Beechgrove. This meant that the primary purpose of the Glencorse trial was to assess the impact of different plant materials, namely grass and clover, on the N cycle following incorporation. The following hypotheses were tested:

a) Nitrogen mineralisation following incorporation of unfertilised, cut grass-clover swards is greater than that from unfertilised, cut grass swards.
b) Gaseous N loss from the soil system following the incorporation of unfertilised, cut grass-clover swards is greater than that from unfertilised, cut grass swards.
c) Nitrogen mineralisation following incorporation of previously arable land is smaller than that from grassland, regardless of sward composition.

The Glencorse trial was also used by Vinten *et al.* (1996). As part of their experiment, these authors required fertiliser to be applied to the CN and PG treatments. In order that a true comparison between treatments was possible after the fertiliser application date, fertiliser should also have been applied to the PC treatment. However, it was felt that the anticipated increase in variability introduced by fertiliser application (section 4.2.2) could mask the effects of clover in the PC treatment and therefore no fertiliser was applied.

Table 3.1.2.4 Treatments applied at the Glencorse field trial, 1993.

Treatment description (Abbreviation)	Plots	Fertiliser (kg N ha^{-1})
Ploughed out cut clover-rich sward, resown to ryegrass (PC)	1, 8	0
Ploughed out cut grass sward, resown to ryegrass (PG)	4, 5	60[b]
Spring barley stubble[a], resown to ryegrass (AN)	2, 3	60[c]
Spring barley stubble[a], resown to ryegrass (CN)	6, 7	60[b]

[a] Stubble was ploughed out in November 1992.
[b] Applied as calcium nitrate.
[c] Applied as ammonium nitrate.

3.1.3 Cowloan

The Cowloan field trial was situated on Boghall Farm, on the Bush Estate, about 12 km south of Edinburgh (NT247653), altitude 200 m. The land capability class for agriculture is 3.1, with wetness limitations (Bibby *et al.*, 1982). The field contains two soil series, Macmerry and Duncrahill Series, both of which are imperfectly drained sandy loam, the latter having a slightly lighter texture due to greater fluvial sorting (Ragg and Futty, 1967). The topsoil properties include: pH 6.3; organic matter content 8.6%; 0.34% total N; further details are given in Appendix 7.

With regard to this thesis, the purpose of the Cowloan field trial was to measure over winter leaching losses after the sowing of an arable crop, rather than grass, following incorporation of the swards (Table 3.1.3). The following hypotheses were tested:

a) Leaching losses following incorporation of grass-clover swards are greater than that from grass swards.
b) Leaching losses following incorporation of grassland swards, regardless of composition, are greater than that following previously arable land.
c) Leaching losses following incorporation of grassland swards, regardless of composition, are greater than that under undisturbed grass-clover swards.

Table 3.1.3 Treatments applied at the Cowloan field trial, 1992.

Treatment description	Abbreviation
Ploughed out 5 year old cut grass-clover ley, sown to spring barley[a]	OGC
Ploughed out 5 year old cut ryegrass ley, sown to spring barley[b]	OPG
Continued arable, sown to spring barley[b]	ARA
Continued undisturbed cut grass-clover sward[a]	CGC

[a] Duncrahill soil series.
[b] Macmerry soil series.

All three ley treatments were ploughed out in late February 1992. Spring barley was drilled on 25 March 1992. No fertiliser was applied. Soil and plant samples were taken on the same occasions at intervals throughout the season to monitor the yield and N uptake of the spring barley crop. After harvest, plots were not ploughed and considerable regrowth occurred. Soil sampling continued and porous cup samplers were installed on 20 October, with samplers also installed in the continued grass-

clover guard area to act as a control. Three porous cups were installed in each plot as described in section 3.7.2 and sampled as described in section 3.7.3. Nitrate-N leaching loads were calculated using drainage data from the adjacent hydrologically isolated plots at the No. 3 field, Bush Estate (Vinten *et al.*, 1992).

3.2 FIELD EXPERIMENTS

3.2.1 Bromide tracer experiment

Given the potential importance of leaching losses in calculating the nitrogen balance for the Beechgrove field trial, it was important to have an independent measure of the effectiveness of porous cups in measuring the leaching of a solute from the soil. Since bromide (Br) is generally found to have very low background levels in soil and its movement in soil is similar to NO_3^--N, the introduction of a Br pulse provides an excellent technique for assessing NO_3^--N measurement techniques (Smith and Davis, 1974). By applying a known amount of solute to a known area which was sampled more intensively than the whole area of the field trial, a quantitative estimate of the performance of porous cups could be made.

An area 4.2 x 7.5 m in plot CGC 1 on the Beechgrove trial was chosen to carry out a bromide tracer experiment. The area contained five porous cups installed at two depths (two at 55 cm, three at 40 cm) (Table 3.7.2) in a non-randomised pattern.

Ten litres of 23.25 g $CaBr_2 2H_2O$ l^{-1} solution, equivalent to 5 g Br m^{-2}, was applied to the area on 8 February 1993 using a backpack sprayer. Four passes were made over the plot to ensure uniform tracer application. Prior to application a 100 ml sub-sample of tracer solution was taken for subsequent analysis.

Porous cups were sampled using the sampling procedure described in section 3.7.3. A soil core was taken to 1 m on 12 October 1993, cut into 20 cm sections, and analysed for Br. Vegetation was not analysed for Br content and so vegetation Br uptake was estimated (Appendix 1.1).

3.2.2 Comparison of soil water nitrate-N concentrations in porous cups, soil cores and drains at Glencorse

Four soil water NO_3^--N concentration sampling techniques were compared over the winter of 1993-4 on plots 4, 7 and 8 at the Glencorse field trial. The four techniques compared were: porous cup sampling (section 3.7.3); deep soil coring (section 3.6.3); drainage from hydrologically isolated plots (section 3.7.4); well samplers (section 3.7.4). Duplicate samples were taken using each technique, apart from the drainage from hydrologically isolated plots of which there was no duplication.

The hypothesis tested was that porous cups provide a reliable and accurate estimate of leaching losses on the Winton soil series. The results of this experiment would allow data collected at the Beechgrove trial to be more critically assessed.

3.3 WEATHER AT THE TRIAL SITES 1992-4

Meteorological data for the whole experimental period was gathered from the Bush House weather station, located within 2 kilometres of all of the trial sites at an altitude of 185 m (section 4.5.1). Given the proximity, similar altitude and location on the same side of the adjacent Pentland Hills, the data from Bush House was considered to be sufficiently representative for all three trial sites. Whilst some meteorological data was available for the individual trial sites, measurements were not taken at a specific time each day. It was therefore considered inappropriate to use this data to attempt any conversion of the Bush House data.

Given the climatic dependence of the processes discussed in this thesis, the results reported can not be extrapolated to other areas without due consideration of changes in meteorological conditions. Furthermore, the meteorological conditions experienced during the two field seasons should not necessarily be regarded as typical for the area (section 4.5.1).

3.4 LABORATORY EXPERIMENTS

3.4.1 Mineral nitrogen release, nitrous oxide and carbon dioxide emissions following incorporation of grazed and ungrazed grass-clover swards

This experiment was carried out in order to aid the explanation of the observed differences in mineral N release and gaseous N losses following sward incorporation in 1992 and 1993 at Beechgrove. The following hypotheses were tested:

a) Nitrogen mineralisation following incorporation of previously grazed swards is greater than that from previously cut swards.
b) Gaseous N loss following incorporation of previously grazed swards is greater than that from previously cut swards.
c) Microbial activity following incorporation of previously grazed swards is greater than from previously cut swards.

Three replicate sampling areas were chosen at random in each of two grass-clover swards at the Beechgrove trial site on 29 June 1994. Until July 1992 both swards, part of the grass-clover holding paddock (section 3.1.1.1), had been grazed each season since 1987. The ungrazed sward, part of the undisturbed grass-clover treatment, had been subject to a cutting regime since July 1992. The grazed sward remained as part of the holding paddock and was subject to grazing according to the requirements for sward height maintenance in the Beechgrove experimental paddocks.

The period of cutting prior to incorporation was greater on the ungrazed sward than on the resown 1993 treatments. However, in 1993 the grazed sward was only infrequently grazed and had to be cut towards the end of the growing season. This decrease in the actual grazing period therefore makes the grazed and ungrazed swards used in the incubation experiment more representative of the swards incorporated on the Beechgrove field trial in 1992 and 1993, respectively. Unfortunately no records of stocking rates in the holding paddocks were kept and therefore a more thorough assessment of their representativeness was not possible.

At each sampling area one turf piece (20 × 20 cm, 0-5 cm depth) and about 3 kg of soil from 5-20 cm depth, was removed for herbage and soil macro-organic matter (MOM) analysis. Larger samples of turf and topsoil were also taken to provide sufficient soil for incubations. Duplicate soil samples, 0-20 cm depth, each consisting

of three augered samples, were taken from locations adjacent to each sampling area, extracted and analysed for NH_4^+-N and NO_3^--N. Five soil samples, 0-20 cm depth, were taken at random from each sward and bulked for determination of gravimetric moisture content.

Soil samples taken for estimation of soil MOM were stored at 4°C prior to extraction. Turf samples were extracted individually. Topsoil samples were thoroughly mixed, subsampled and bulked to give one 3 kg sample for each sward. Prior to extraction, large stones (>27 mm dimension) were removed.

Topsoil samples for incubation were sieved through a 27 mm mesh to remove large stones and increase sample homogeneity. Turf samples were chopped into *ca.* 4 cm^3 pieces and thoroughly mixed with the topsoil. Three replicate buckets, 22 cm inner diameter and 18 cm depth, were loosely filled (Dry Bulk Density = 0.84) with *ca.* 3.44 kg of this mixed soil. This procedure was repeated for each of the six sampling points.

Sub-samples of the mixed soil were taken immediately for determination of gravimetric moisture content and mineral N on day 0 of the incubation period. Sub-samples were also taken for soluble organic carbon (SOC) analysis, having been stored at 4°C for eight days.

On day 2 of the incubation period, each bucket was wetted up with distilled water to a gravimetric moisture content of 0.31 g H_2O g dry $soil^{-1}$, equivalent to 38% water-filled pore space (WFPS-Equation 3.4.1). Soil was subsequently kept at this moisture content by weighing and rewetting every 2-3 days until day 47 when soil gravimetric moisture content was increased to 0.35 g H_2O g dry $soil^{-1}$, equivalent to 43% WFPS. Distilled water was applied using a fine mist sprayer.

Since the bulk density of the chopped turf pieces was unlikely to have changed significantly from their field intact state, the values for WFPS were likely to be higher than that of the surrounding soil.

Equation 3.4.1

$$\text{Water} - \text{filled pore space} = \frac{\theta_g \times \text{DBD}}{P}$$

where:

θ_g = soil gravimetric moisture content (g H_2O g $soil^{-1}$)

DBD = soil dry bulk density (g cm^{-3})

P = total soil porosity = (1-(DBD/particle density))

where particle density = 2.65 g cm^{-3}

One replicate bucket of soil from each sampling area was sampled for soil (SOIL) on days 8, 14, 21, 28, 39 and 46 of the incubation period. Bulk samples of *ca.* 60g were taken from two points within each replicate and analysed for mineral N. The other two replicate buckets (GAS and ACET) were both sampled for CO_2 and N_2O on the first nine days of the incubation period and then on days 10, 11, 13, 15, 18, 21, 25, 28, 33, 39, 43, 47,49. On days 11, 18, 28, 39 and 49 ACET replicates had acetylene (C_2H_2) applied to inhibit the transformation of N_2 to N_2O. On day 53 all buckets were emptied, thoroughly mixed, subsampled and frozen for subsequent mineral N and SOC analysis.

Buckets were incubated at approximately outside air temperature and arranged in two columns, three rows and three heights as a paired, Latin square design. The air-tight bucket lids had been adapted to allow gas sampling through Subaseals.

Gas samples were taken from GAS and ACET replicate buckets 90-180 minutes after their lids had been sealed. Duplicate 5 ml and 1 ml samples were taken for N_2O and CO_2 analysis, respectively, and analysed as described in section 3.8.4.

When C_2H_2 was to be applied to ACET replicates 20% of the headspace air was removed and replaced with C_2H_2 (section 3.8.3). Acetylene was then allowed to diffuse into the soil for eight hours. After eight hours lids were removed for one minute, to flush the headspace, and replaced. Time zero gas samples were taken from each individual bucket. The $N_2 + N_2O$:N_2O ratios were calculated using estimates of the N_2O flux from ACET replicates (had C_2H_2 not been applied) which were calculated using the N_2O emission ratio between the GAS and ACET replicates on the nearest sampling date when neither replicate had C_2H_2 applied to it.

On all occasions lids were sealed on every bucket for exactly the same time period.

3.5 PLANT SAMPLING, PREPARATION AND ANALYSIS

3.5.1 Residue inputs

3.5.1.1 Beechgrove trial

In 1992, plant top samples were taken on 27 May and 2 June from the grass-clover and grass paddocks, respectively, to estimate the clover content of both swards prior to ploughing. Plant top samples were also taken on 11 June to estimate the amount of plant top residues ploughed in, its clover content and N content. Root mass was estimated by taking six replicate turf pieces (18 × 18 cm, 0-4 cm depth) from each site prior to ploughing. Root matter in the 20-40 cm layer was estimated using deep soil cores taken from undisturbed treatments on 18 June.

In 1993, residue inputs were estimated more thoroughly. Sward clover contents were attained from plant top samples taken on 4 May and 6 June. On 31 March, duplicate turf samples were taken from each plot of the resown 1993 treatments (PGC93 and PG93). Soil cores were also taken to a depth of 40 cm, from where the turf had been removed (Table 3.5.1.1). All soil samples were frozen and subsequently extracted for MOM as described in section 3.5.3.2.

Table 3.5.1.1 Residue input sampling at the Beechgrove trial, 1993.

Residue definition	Height / Depth (cm)	Area sampled (cm^2)
Plant tops	above *ca.* 4	5000
Stubble (Above-ground)	0-*ca.* 4	400
Turf MOM[a]	0-4	400
Soil MOM[b]	4-40	*ca.* 34

[a] Below-ground.
[b] This was split into three sections: 4-10, 10-20 and 20-40 cm.

3.5.1.2 Glencorse trial

Residue inputs were estimated using exactly the same procedure as at Beechgrove in 1993. Only PC (ploughed out clover-rich sward) and PG (ploughed out grass sward) treatments were sampled.

3.5.2 Vegetation uptake of nitrogen

3.5.2.1 Beechgrove trial

Herbage sampling took place approximately once a month throughout the growing season (Table 3.5.2.1). Herbage was cut to a sward height of *ca.* 4 cm. One sample, 0.5 m^2, was taken from each plot during 1992. In 1993 duplicate cuts were taken from each plot to increase the accuracy of herbage N uptake estimates.

Table 3.5.2.1 Dates of plant top sampling on the Beechgrove trial, 1992-3.

Date	Treatments sampled
7 August 1992	Undisturbed swards, resown 1993 treatments
16 September 1992	All [a]
15 October 1992	All
30 March 1993	All [a]
4 May 1993	All [a]
6 June 1993	All bar resown 1993 treatments
9 August 1993	All [b]
17 August 1993	PGC93
22 October 1993	All [a]
17 December 1993	All [a]

[a] except fallow treatments.
[b] except PGC93 and fallow treatments.

The roots of the resown 1992 treatments were not sampled at the end of the 1992 growing season. These roots may have contained a substantial amount of N which would not be available for leaching. An estimate of root growth was calculated based on the above-ground herbage DM production (Appendix 2.7).

3.5.2.2 Glencorse trial

Grass establishment was poor at Glencorse in 1993 and vegetation samples were only taken on 21 March 1994. Four replicate samples of herbage tops, 25 × 25 cm, were cut to the soil surface in each plot. Duplicate soil samples, *ca.* 34 cm^2, were taken to a depth of 10 cm from plots 1 and 2 to estimate root uptake of N. These plots were representative of the poorest and best grass establishment at the site, respectively.

3.5.3 Plant preparation and analysis

3.5.3.1 Plant tops

All herbage samples were oven dried at 100°C and DM recorded. Samples were coarsely milled in a hammer mill and then sub-samples (3-5 g) were finely ground in an agate ball mill to achieve adequate homogeneity in the very small samples taken for N analysis (Robinson and Smith, 1991), and to allow analysis for total N content. All plant fractions were assumed to have a C content of 45%.

3.5.3.2 Plant roots and macro-organic matter

An exact quantification of plant roots under established swards is extremely difficult (Williams and Baker, 1957). Instead soil MOM was extracted by dispersing soil samples overnight in a 5% sodium hetamexaphosphate solution. The solution was stirred and soil particles allowed to settle. The solution was then decanted through a 425 μm sieve and MOM gently washed whilst on the sieve. This procedure was repeated about 10 times until all soil particles were removed.

This technique substantially overestimates plant root mass in the resown swards since it includes undecomposed MOM remaining since ploughing. However, this method does provide an adequate quantification of the mass of residues incorporated.

3.5.3.3 Sward clover content

Dried herbage samples were hand-sorted for clover and figures are given as a percentage of total DM, unless otherwise stated. Initially sub-samples from each plot were sorted for each sampling date. However, this proved far too time consuming. Subsequently only samples taken prior to ploughing were sorted, when accurate estimates of residue inputs were required. For other sampling dates plot sub-samples were bulked to give a treatment sample or, an estimate was used (Appendix 2.4).

Estimation of sward clover content, by ground cover, was carried out using quadrat analysis on 27 May 1992. A 1 m^2 quadrat was divided into 10 × 10 cm squares. Clover percentage ground cover was estimated for each of these sections in two randomly chosen quarters of the large quadrat. Six replicate quadrat analyses were carried out on each sward, completed in random order to avoid bias on either sward.

3.6 SOIL SAMPLING, PREPARATION AND ANALYSIS

Soil samples were taken at intervals during the growing season in all the field trials (1992-4). Soil was sampled to 20 cm depth from random positions in the plots, using a Dutch auger. Exact sampling procedures varied and are given in sections 3.6.1 and 3.6.2. Larger soil cores taken for cup installation (section 3.7.2) and deep soil sampling (section 3.6.3) provided additional samples.

3.6.1 Beechgrove trial

The dates on which soil was sampled are given in Table 3.6.1. In 1992 soil was sampled less frequently than on a weekly basis following initial ploughing and rotavation. A clearer N release pattern may have been detected if weekly sampling had been adopted shortly before ploughing, with less frequent sampling towards the end of the growing season (September-October) when sampling was unnecessarily intense. Sampling frequency and timing was consequently adjusted in 1993.

Initially only one bulk sample of two cores was taken from each plot, and only one extraction carried out on each sample. This sampling and analysis framework frequently produced unanticipated and excessively variable soil mineral N data. An improved framework was adopted, as of 31 August 1992, to attain a more accurate estimate of the true plot means. Two samples were taken from each plot, consisting of two cores bulked. Two extractions were carried out on each sample.

Table 3.6.1 Soil sampling dates at the Beechgrove trial, 1992-3.

Treatments sampled (Sampling depth, cm)	Date
CGC, PGC93, CG, PG93 (0-100)	17 & 18 June
CGC, PGC92, CG, PG92 (0-20)	2 July
CGC, PGCF, CG, PGF (0-20)	14 July
All except PGC93 and PG93 (0-20)	23 July
All except PGC93 and PG93 (0-20)	28 July
All except PGC93 and PG93 (0-20)	7 August
All except PGC93 and PG93 (0-20)	14 August
All except PGC93 and PG93 (0-20)	21 August
All except PGC93 and PG93 (0-20)	31 August
All except PGC93 and PG93 (0-20)	15 September
All except PGC93 and PG93 (0-20)	29 September
All except PGC93 and PG93 (0-20)	5 October
All (0-100)	12 & 13 October
All except PGC93 and PG93 (0-20)	27 October
All except PGC93 and PG93 (0-20)	20 November
All except PGC93, CGC, PG93 and CG (0-100)	21 January
All (0-20)	25 February
All (0-20)	12 March
All (0-20)	30 March
All (0-20)	8 April
All (including EP from this date onwards) (0-20)	15 April
All (0-20)	20 April
All (0-20)	30 April
All (0-20)	11 May
All (0-20)	19 May
All (0-20)	26 May
All (0-20)	2 June
All (0-20)	10 June
All (0-20)	17 June
All (0-20)	24 June
All (0-20)	1 July
All (0-20)	8 July
All (0-20)	15 July
All (0-20)	21 July
All (0-20)	28 July
All (0-20)	4 August
All (0-20)	16 August
All (0-20)	25 August
All (0-20)	2 September
All (0-20)	9 September
All (0-20)	21 September
All except PGC92 and PG92 (0-100)	11 & 12 October
All (0-20)	3 November

3.6.2 Glencorse trial

Three soil samples, each consisting of two cores bulked, were taken from all plots on the dates shown in Table 3.6.2.

Table 3.6.2 Soil sampling dates at the Glencorse trial, 1993-4.

Sampling Depth (cm)	Date
0-20	29 June 1993
0-20	19 July
0-20	26 July
0-20	4 August
0-20	12 August
0-20	19 August
0-20	26 August
0-20	2 September
0-20	9 September
0-20	20 September
0-20	8 October
0-100[a]	20 October
0-100[a]	19 January 1994

[a] Only plots 4, 7 and 8 sampled. Two cores per plot (Section 3.6.3).

3.6.3 Deep soil coring

Deep soil cores were used to quantify N, C and roots in the soil profile at the start of the Beechgrove trial and to supplement porous cup data on NO_3^--N leaching.

Cores were taken to a depth of 1 m with a percussion corer, 6.6 cm diameter. On each occasion, profile details were noted and cores were cut into 20 cm sections. The whole of each core section taken at Beechgrove on 17 and 18 June 1992 was kept to enable quantification of MOM, but on all other occasions cores were subsampled in the field prior to storage. One core was taken from each plot, unless otherwise stated. Details of sampling dates, and treatments sampled, are given in Tables 3.6.1. and 3.6.2.

3.6.4 Soil preparation and analysis

3.6.4.1 Routine analysis

All soil samples were immediately sealed in plastic bags to prevent moisture loss and were stored at 5°C overnight, where analysis was to occur the next day. Otherwise soil samples were stored frozen (-15°C) until analysis could be carried out. Soil pH and organic matter determinations were carried out on soil samples from the field trial sites at the beginning of the season and soils used in the laboratory and field experiments were fully characterised before use. Soil texture was determined by particle size analysis (Gee and Bauder, 1986). Soil pH was determined in water (McLean, 1982). Soil organic matter was determined by the Walkley-Black method (Allison, 1965). Extractable P was determined by colorimetry, extractable K by flame photometry and extractable Mg by atomic absorption spectrophotometry (MAFF, 1986).

3.6.4.2 Mineral nitrogen determination

Soil samples were thoroughly mixed and two sub-samples taken to determine the soil gravimetric moisture content, by drying in an oven at 105°C for 24 hours, or until a constant weight was attained. Unless otherwise stated, two sub-samples of *ca.* 20 g of fresh soil were accurately weighed into separate 250 ml flasks and each shaken with 100 ml of 1M KCl extracting solution for one hour. The extractant was then filtered (Whatman No. 42 filter papers) and NH_4^+-N and (NO_3^- and NO_2^-)-N determined by continuous flow analysis (Crooke and Simpson, 1971; Best, 1976) using a Chemlab autoanalyser.

3.6.4.3 Water soluble organic carbon (SOC) determination

Soil samples were thoroughly mixed and three sub-samples taken to determine the soil gravimetric moisture content (section 3.6.4.2). Three sub-samples of *ca.* 40 g of fresh soil were accurately weighed into separate 250 ml flasks and each shaken with 80 ml of distilled water for twenty minutes. The extractant was then centrifuged for twenty minutes at 3500 rpm and *ca.* 3 ml was filtered again using a Millipore Millex-LCR 0.5 μm filter unit.

One drop of orthophosphoric acid was placed in each filtered extract, which had carrier gas bubbled through it ("sparged") for approximately five minutes to remove

inorganic C as CO_2. Soluble (non-purgeable) organic C concentrations, from 200 µl sub-samples, were analysed on a Rosemount Dohrmann DC-80 total organic C analyser (Sartec Limited, Kent). Carbon content was calculated by ultra-violet promoted persulphate oxidation, followed by infrared detection of the sparged CO_2. This was calibrated using a 400 ppm C standard in potassium hydrogen phthalate.

3.7 SOIL SOLUTION SAMPLING AND ANALYSIS

3.7.1 Porous cup specifications

Porous cups were supplied by Van Walt Ltd., Haslemere, Surrey and produced by Nardeux Humisol, St.Avertin, Cedex, France. Samplers were of the "cane" design and consisted of a 60 cm length of hollow PVC pipe, 6.3 cm in diameter, cemented onto a ceramic cup 6.3 cm in diameter and 7.5 cm in length. A rubber bung, with copper piping and flexible tubing inserted through its midpoint, provided a point for evacuation of the system. A pinch valve on the tubing gave an air-tight seal in order to maintain the applied suction.

3.7.2 Porous cup preparation and installation

All porous cups were washed with 0.1M HCl and thoroughly rinsed with distilled water prior to installation.

Soil cores, 6.6 cm in diameter, were taken to the depth of cup tip placement (40 cm below the soil surface for all cups except C18 and C1 which were installed at 55 cm depth). Soil cores were divided into 20 cm sections and, having set aside sufficient backfill material, stored for mineral N analysis. A Dutch auger was used to remove a further 5 cm of soil to ensure silica would remain under the cup tip once installed. This soil was kept for particle size distribution analysis. Cups were installed a few days after soil core removal to enable any possible soil smearing to dry, and reopen.

Approximately 450 cm^3 of silica slurry mixed to a stiff consistency, but moist enough to allow penetration of the whole cup, was poured into the base of the hole. The cup was then quickly and carefully pushed into the slurry to the correct depth. Slurry rose up around the side of the cane, to 20-27 cm below the soil surface (Table 3.7.2), encasing the porous surface. Bentonite powder was then tamped down on top of the silica to prevent preferential flow down the side of the cane. The remainder of the hole was backfilled to the surface with soil from the appropriate depth.

At Beechgrove one cup was installed in each plot to give an indication of inter-plot variability. Triplicate cups were installed, approximately 2 m apart, in plots PGCF 3, PGC92 2 and CGC 1 in order to appraise intra-plot variability.

At the Glencorse trial duplicate cups were installed in plots 4,7 and 8 on 7 and 8 September 1993.

3.7.3 Porous cup sampling procedure

The triplicate cups in plots PGCF 3, PGC92 2 and CGC 1 were used to determine a suitable sampling procedure. During August and September 1992 the sampling duration of each replicate was varied between 1 and 4 days to assess how long a -20 kPa suction could be maintained, and thus how long soil solution was actually being sampled. Between 13 August and 9 October 1992 samples were taken from cups using -20 or -70 kPa suction in order to test the effect of applied suction on the NO_3^--N concentration of the sampled soil solution. Whilst samples gathered using the two suction levels were not necessarily taken on the same days (sampling dates may have been up to three days apart), given the resources available and the variation between cups within treatments (section 5.5), this was the most appropriate comparison possible. Whilst replication was limited, results suggested that the NO_3^--N concentration of the sampled soil solution was not affected by the suction applied (Appendix 3). A low suction, -20 kPa, two day sampling period strategy was chosen.

Porous cups were sampled weekly when the soil was sufficiently moist to yield a sample. Samples were collected by lowering a section of narrow bore PVC tubing to the base of the cup and applying a suction, via a polyethylene sampling bottle. This allowed sampling of very small volumes (Wagner, 1962). Samples were weighed to calculate their volume.

Table 3.7.2 Porous cup installation details at Beechgrove.

Plot	Cup Number	Installation Date	Silica depth (cm)[a]	Bentonite Depth (cm)[a]
PGC92 1	C5	7 October 1992	25	9
PGC92 2	C22	30 July 1992	25	14
PGC92 2	C24	30 July 1992	25	5
PGC92 2	C28	30 July 1992	19	8
PGC92 3	C36	7 October 1992	28.5	7.5
CGC 1	C20	29 July 1992	23	13
CGC 1	C21	29 July 1992	21	10
CGC 1	C23	29 July 1992	23	15.5
CGC 2	C32	6 October 1992	22	9
CGC 3	C30	6 October 1992	25.5	9
PGCF 1	C12	7 October 1992	ND	ND
PGCF 1	C50[b]	27 September 1993	20	6
PGCF 2	C7	8 October 1992	36	13.5
PGCF 3	C25	30 July 1992	27	13
PGCF 3	C26	30 July 1992	19.5	11
PGCF 3	C27	30 July 1992	24.5	11.5
PG92 1	C11	28 September 1992	21	10
PG92 2	C33	29 September 1992		
PG92 3	C2	1 October 1992	17	8
CG 1	C31	1 October 1992	26.5	6
CG 2	C38	1 October 1992	25.5	6
CG 3	C34	6 October 1992	25	7.5
PGF 1	C10	24 August 1992	28	13
PGF 2	C29	24 August 1992	25	13
PGF 3	C3	24 August 1992	27.5	11
PGC93 1	C6	29 June 1993	16	5
PGC93 2	C4	2 July 1993	22	13.5
PGC93 3	C15	2 July 1993	15	4
PG93 1	C16	28 June 1993		
PG93 2	C19	28 June 1993	21	11
PG93 3	C37	29 June 1993	25	16
CGC 1	C18	10 August 1992	40	32
CGC 1	C1	10 August 1992	41.5	32.5

[a] Depth referenced to the soil surface.
[b] Cup 12 was smashed during ploughing in 1993, and replaced by cup 50.
ND = Not determined.

3.7.4 Drainage sampling and hydrological measurements at Glencorse

Rainfall was measured using two tipping bucket rain gauges mounted on concrete plinths adjacent to the plots. Drainage from individual plots was piped to tipping bucket flow meters (Field Drainage Experimental Unit, Anstey Hall, Trumpington, Cambridgeshire) and tips were logged by electronic integral counters. Flow weighted samples (Vinten *et al.*, 1991) were collected approximately once a week.

In addition to the flow weighted drainage samples, in August 1993 open dip wells were installed on plots 4, 7 and 8. These were created by removing a soil core to a depth of 1 m using a percussion corer, 6.6 cm in diameter. These were sampled approximately weekly over the winter of 1993-4.

3.7.5 Mineral nitrogen analysis of water samples

Water samples were stored at 4°C and analysed within fourteen days for NH_4^+-N and NO_3^--N by continuous flow analysis (Crooke and Simpson, 1971; Best, 1976).

3.7.6 Bromide analysis of water samples

Bromide was analysed using the potentiometric method with an ion selective electrode. 0.5, 1, 5 and 10 mg $Br^- l^{-1}$ standards were used to calibrate electrode readings. 0.5 ml ISA (1M NaOH) was added to 25 ml of sample or standard to give ionic balance (Adriano and Doner, 1982). Samples and standards were then stirred for 50 seconds on a magnetic stirrer and then analysed. The Br^- content was related logarithmically to the mV reading.

3.7.7 Calculation of solute leaching loads on the Beechgrove and Glencorse field trials

Solute leaching loads for the bromide tracer experiment, Beechgrove and Glencorse field trials were calculated using drainage data from the hydrologically isolated plots at Glencorse (section 3.7.4). The two field sites were less than 1 km apart, with little difference in altitude (sections 3.1.1.1 and 3.1.2.1). No attempt was made to account for possible differences in evapotranspiration between the treatments given their relative size compared to the substantial overwinter drainage.

The use of field drainage installations such as those at Glencorse is discussed in section 2.2.5.2. Given the similar soil types at the Glencorse and Beechgrove sites, the calculated drainage volumes from Glencorse (Table 4.5.2) were considered to be the easiest and most reliable data set to use. The alternative techniques would have involved a great deal of additional work and are generally considered less appropriate on structured clay soils (section 2.2.5.2). The validity of the use of Glencorse drainage data at Beechgrove is discussed further in section 4.1.5.1.

On the Beechgrove trial in 1992 the staggered installation of porous cups (Table 3.7.2) meant that NO_3^--N concentrations in some plots had to be predicted between 14 August and 5 October. Various methods were used to estimate the missing soil water NO_3^--N concentration data. Firstly, soil NO_3^--N data from deep cores and topsoil samples in plots missing cup data were compared with the respective data in plots where cup NO_3^--N concentrations were measured. This allowed prediction of timing and concentration of peak in soil water NO_3^--N. Secondly, ratios of soil water NO_3^--N concentrations between plots within treatments were calculated for dates when all plots yielded cup samples. The ratios from the nearest sampling dates were then used to predict soil water NO_3^--N concentrations in unmeasured plots where cup data was only available for one or two plots.

By the time the final cups were installed as much as 17% of the drainage for the 1992-3 drainage season had occurred. The quantitative inaccuracy of calculated treatment mean NO_3^--N loads incurred due to the predicted data was probably greatest in the PGC92 treatment, followed in descending order by the PG92, PGCF, PGF, CGC and CG treatments.

On dates where individual cups failed to yield samples, NO_3^--N concentrations were estimated by assuming a linear pattern between measured sampling dates. Given the smooth nature of NO_3^--N concentration curves for individual cups (Appendix 4), this was suitable on the majority of occasions. However, where this method was considered to be inappropriate, NO_3^--N concentrations were estimated according to trends shown in NO_3^--N concentration curves of measured cups within the relevant treatment whose curves' trends matched that of the predicted cup.

3.8 GASEOUS NITROGEN SAMPLING

Gaseous N emissions were measured in the field using closed chambers of two different designs. Design A consisted of a 20 cm length of polypropylene pipe (diameter 40 cm) fitted with a 4.5 cm wide outward-facing PVC flange (Smith *et al.*, 1995). This was sealed using a square of 3 mm aluminium sheet fitted with a circle of rubber draught excluder on its underside, clipped tightly onto the flange. Design B consisted of 70 × 70 cm steel frames, 15 cm deep with a 3 cm wide inward facing flange (Arah *et al.*, 1991; Figure 3.8a). Aluminium lids fitted with draft excluder were weighted onto the flange using metal bars.

Gas samples were taken at the beginning and end of each incubation period, using 5 ml air-tight greased syringes, in order to calculate N_2O fluxes using equation 3.8.1.

3.8.1 Beechgrove trial

During July and August 1992 measurements of N_2O fluxes were carried out using chamber design B, on four treatments: CGC (2 replicates), PGCF (4 replicates), CG (2 replicates) and PGF (2 replicates). Chambers were sealed for about 30 minutes. The timing of sampling varied from about 10 a.m. to 3 p.m. and as a result may have introduced complications caused by diurnal fluctuations in flux rates (Denmead *et al.*, 1979). This work was carried out as part of an M.Sc. thesis (Bergos, 1992).

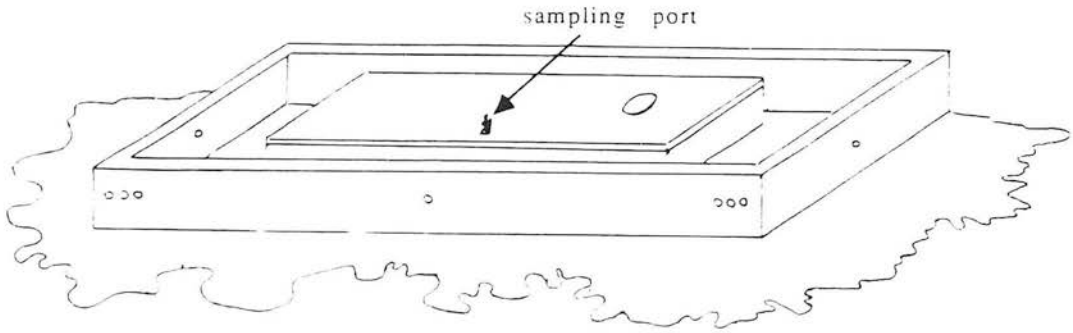
Equation 3.8.1

$$\text{Nitrous oxide flux (g } N_2O\text{ - N ha}^{-1}\text{ day}^{-1}) = \frac{c \times h \times 168}{t}$$

where:
c = change in concentration of N_2O (ppm)
h = height of chamber lid above the soil surface (cm)
t = length of time chamber was sealed (minutes).

In 1993 a more comprehensive study of N_2O fluxes was carried out. Duplicate chambers, of design A, were installed in each plot of the resown 1993 treatments. Duplicate chambers, of design B, were installed in plot EP. One replicate chamber, design B, was installed in each plot of the remaining treatments. Sampling time was standardised to between 1 p.m. and 4 p.m.

(a) Flux measurement



(b) Acetylene treatment

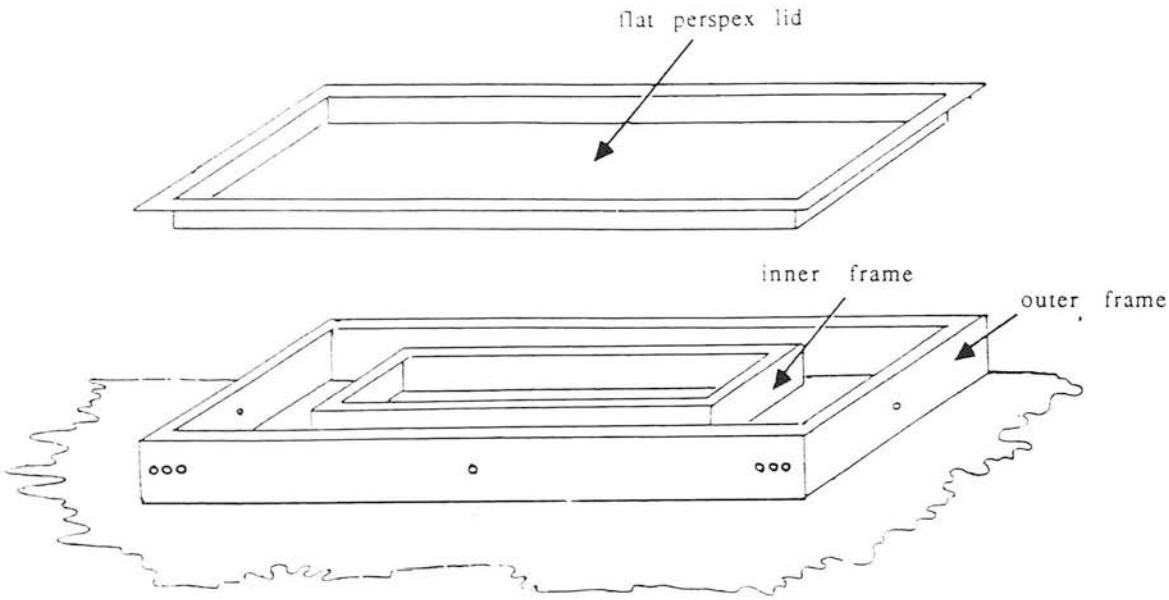


Figure 3.8 Chambers for field application of acetylene and measurement of gaseous emissions from soils: (a) View showing perspex lid for covering the soil during application of acetylene; (b) Design B closed chamber. (Modified version of that described by Arah *et al.*, 1991).

3.8.2 Glencorse trial

Duplicate chambers (design A) were installed in both plots of treatments PC (ploughed out clover-rich sward) and PG (ploughed out grass sward) and, in plots 7 and 2 of treatments CN and AN, respectively.

3.8.3 Acetylene inhibition

In both years of the Beechgrove trial denitrification fluxes were occasionally measured using the C_2H_2 inhibition technique.

Larger chambers were placed over the existing chambers, which were left open (Figure 3.8b). Acetylene (in the form of gas dissolved in acetone), which had been passed through concentrated hydrochloric acid and distilled water to remove acetone (Walter *et al.*, 1979) was passed into the surrounding chamber for 90-120 minutes, allowing time for its diffusion into the soil profile, to inhibit the reduction of N_2O to N_2 (Ryden *et al.*, 1979). Replicate chambers were sampled without the use of C_2H_2 in order to calculate the proportions of denitrification products, N_2O and N_2. Denitrification fluxes were only measured on the grass-clover block due to equipment limitations.

3.8.4 Gas analysis

Gas chromatographic methods were used to determine gaseous emissions of N_2O, CO_2 and C_2H_2. N_2O was analysed using one of two gas chromotagraphs attached to electron capture detectors (ECDs): (i) a Hewlett Packard 5890 with a "Porapak" Q column. Detector and oven temperatures were 380°C and 55°C, respectively. (ii) Pye Unicam 104 fitted with a Tracor ECD and linearizer and a "Porapak" Q column. Detector and oven temperatures were 380°C and 40°C, respectively. All gas samples sent to the ECD for N_2O analysis were backflushed to prevent any gas with a column retention time longer than N_2O from being analysed. A Pye Unicam 104 gas chromatograph fitted with a thermal conductivity detector was used for analysis of CO_2 and C_2H_2 samples. Detector and oven temperatures were 120°C and 70°C, respectively.

3.9 MASS SPECTROMETRIC ANALYSIS OF SOIL AND PLANT SAMPLES FOR NITROGEN

The technique of mass spectrometry determines the isotopic composition of a sample by separating charged ions on the basis of their mass to charge ratio and determining their relative proportions (Robinson and Smith, 1991). In the case of N analysis this involves the conversion of N in the sample to N_2. The N_2 molecules are ionised and then passed through a magnetic field which separates the N into three components: $^{14}N_2$, $^{14}N^{15}N$ and $^{15}N_2$, with masses of 28, 29 and 30 respectively. Total N was calculated by addition of the three components.

Plant and soil extracts were analysed for N content using a single inlet VG Isogas MM622 triple detector mass spectrometer linked to a Carlo-Erba 1400 automatic N analyser, which converts N compounds to N_2 by the Dumas oxidation-reduction procedure (Robinson and Smith, 1991).

Sub-samples of the prepared plant material (section 3.5.3.1) and ball-milled soil were accurately weighed out into small tin cups and sealed for analysis. In this system at least 100 μg of N is required to give an accurate reading. For young plant samples with N contents greater than 1.5% N, a sample of 10 mg is sufficient. More mature plant samples and the soil extracts normally require sub-samples of 20-30 mg to ensure that there is sufficient N present for analysis. Reference values are obtained from standards of known N content in the same batch as the samples. These standards allow the mass spectrometer computer software to calibrate the ion beam currents obtained for the samples and calculate values for total percentage N.

Checks carried out have shown that the variability between replicate 5 g sub-samples of plant material selected for grinding in the agate ball mill, and that between replicate 10 mg portions of the subsequently ground material was very much less than is commonly observed between replicate field plots (Robinson and Smith, 1991).

3.10 STATISTICAL METHODS

The data from each experiment were first tabulated in MINITAB and treatment means with accompanying standard errors calculated. Secondary measurements (e.g. crop N uptake) were then calculated for each replicate from the measured data, and these were also tabulated and means and standard errors calculated.

Analyses of variance were carried out for each experiment, where the exact form of the effects and interactions varied according to the experiment. Analysis was carried out in MINITAB and the null hypothesis was tested:

"The differences seen here are no greater than those which would have been seen, if all the plots had been treated identically" (Dyke, 1974).

This enabled the identification of treatments which resulted in N cycle process rates significantly different from the control treatments and perhaps from other treatments applied. These effects were examined with respect to the original hypothesis tested by the experiment, and this original hypothesis was either accepted or rejected.

Unless otherwise stated, treatment errors shown on figures and in tables are always the standard error of the mean (SE). On some of the figures error bars are not given for each individual data point because efforts to do so considerably reduced the clarity of the figure. Coefficients of variation changed through time and so:

a) A single summary statistic on figures was not possible
b) Statistical tests comparing treatments had to be carried out for each individual sampling occasion.

In order to give an indication of errors through time on figures, various techniques have been used and the reader is referred to notes on figures for details. Where statistical differences between treatments exist, these are referred to in the text.

4 RESULTS

4.1 BEECHGROVE FIELD TRIAL

4.1.1 General soil analysis

In the undisturbed treatments organic matter content in the 0-20 cm layer (3.71-4.56% C) was always higher than that in the 20-40 cm layer (1.55-2.89% C) ($p<0.01$). In June 1992, organic matter content in the 20-40 cm layer of the undisturbed grass-clover treatment was significantly greater than that in the undisturbed grass treatment ($p<0.05$), but in October 1992 this pattern was reversed ($p\leq0.001$).

All treatments ploughed in June 1992 showed a decrease in organic matter content in the 0-20 cm layer when sampled in October 1992 (3.64-3.9% C). At least part of this decrease may have been due to 'dilution' by soil of lower organic matter content from 20-30 cm depth. The decrease was not significant and there were no significant differences between any of the treatments at either soil depth.

Total N concentrations in the 0-20 cm soil layer were 0.21% (0.01) and 0.19% (0.01) under the grass and grass-clover swards, respectively, and showed no change during the trial.

4.1.2. Sward residue inputs on the Beechgrove trial 1992-3

In 1992 the total plant residue DM input from the 0-40 cm soil depth and plant tops on the two sward types were not significantly different (Table 4.1.2.1). The grass-clover sward had significantly greater DM input ($p<0.01$), but significantly lower N content ($p<0.01$), in plant tops than the grass sward. There was no significant difference in N inputs from plant tops between sward types. The grass sward had a significantly greater N input ($p<0.01$) from the 0-4 cm horizon than the grass-clover sward, but the respective DM inputs were not quite significantly different ($0.05<p<0.1$).

Table 4.1.2.1 Mean dry matter quantity and N content of sward residues for the Beechgrove trial, 1992 (Standard errors in parentheses[a]).

Residue	Sward	DM (tonnes ha^{-1})	Residue N content (% N)	Residue N input (kg N ha^{-1})
Plant tops[b]	CGC	1.49 (0.08)	2.35 (0.29)	37.1 (5.4)
	CG	1.17 (0.06)	3.55 (0.15)	42.7 (3.7)
Stubble + turf MOM, 0-4 cm	CGC	11.96 (0.72)	1.27 (0.04)	152.1 (8.7)
	CG	14.31 (0.94)	1.44 (0.10)	204.6 (11.8)
MOM, 4-10 cm[c]	CGC	2.13 (0.02)	1.40 (0.03)	29.7 (0.7)
	CG	2.16 (0.19)	1.41 (0.05)	30.3 (2.2)
MOM, 10-20 cm[c]	CGC	1.81 (0.15)	1.29	23.3 (1.9)
	CG	1.50 (0.10)	1.29	19.3 (0.9)
MOM, 20-40 cm	CGC	1.37 (0.16)	1.29[c]	17.7 (2.1)
	CG	1.19 (0.14)	1.29[c]	15.3 (1.8)
Total	CGC	18.77 (1.12)		259.9 (18.7)
	CG	20.33 (1.43)		312.1 (20.4)

[a] For 'plant tops' and 'stubble + turf MOM', d.f.=5. For all other residues d.f.=2.
[b] This does not include plant tops cut right to the soil surface, only that above the cutting height *ca.* 4 cm.
[c] Data is from 31 March 1993.

In 1993 the total plant residue DM input from the 0-40 cm soil depth and plant tops (Table 4.1.2.2) on the two sward types were not significantly different. The grass sward had a significantly higher N input from plant tops than the grass-clover sward ($p<0.05$), but the respective DM inputs were not quite significantly different ($0.05<p<0.1$). There were no significant differences between the two sward types MOM inputs in any soil horizon. Details of clover residue inputs from both sward types, in both years, are given in section 4.1.6.

Table 4.1.2.2 Mean dry matter quantity and N content of sward residues for the Beechgrove trial, 1993 (Standard errors in parentheses, d.f.=2).

Residue	Sward	DM (tonnes ha^{-1})	Residue N content (% N)	Residue N input (kg N ha^{-1})
Plant tops[a]	PGC93	2.18 (0.15)	2.01 (0.05)	45.0 (3.7)
+ stubble	PG93	2.82 (0.22)	2.16 (0.08)	65.5 (2.9)
MOM,	PGC93	8.87 (0.18)	1.4[b]	124.2 (2.5)
0-4 cm	PG93	9.34 (0.69)	1.4[b]	130.8 (9.6)
MOM,	PGC93	2.13 (0.02)	1.40 (0.03)	29.7 (0.7)
4-10 cm	PG93	2.16 (0.19)	1.41 (0.05)	30.3 (2.2)
MOM,	PGC93	1.81 (0.15)	1.29[c]	23.3 (1.9)
10-20 cm	PG93	1.50 (0.10)	1.29 (0.05)	19.3 (0.9)
MOM,	PGC93	1.13 (0.09)	1.29[d]	4.6 (1.2)
20-40 cm	PG93	1.47 (0.25)	1.29[d]	19.0 (3.2)
Total	PGC93	16.12 (0.33)		236.8 (6.5)
	PG93	17.29 (1.36)		264.9 (17.5)

[a] Includes an estimate of grass growth since the previous cut on 4 May 1993, assuming a linear rate of DM production between the cuts on 4 May and 6 June 1993.
[b] Assumed value, based on the data for MOM in the 4-10 cm section.
[c] Assumed to be the same value as for PG93.
[d] Assumed to be the same value as for the MOM in the 10-20 cm section.

4.1.3 Soil mineral nitrogen in the Beechgrove field trial

In 1992 no soil samples were taken prior to ploughing. Samples from the undisturbed treatments taken on 18 June 1992 were used as pre-ploughing control samples. Until grass seed was sown on 20 July 1992 the resown 1992 and fallow treatments were not sampled separately. Thus discussion of the immediate effect of ploughing applies to both these treatments.

Once the initial reactions to ploughing had diminished, temporal patterns in NH_4^+-N concentrations were generally consistent across treatments in both sward blocks

(Figure 4.1.3.1)[1]. In 1993, NH_4^+-N concentrations fluctuated across a generally wider range and treatment differences became more apparent. In 1992 there was no significant effect of sward type on NH_4^+-N, whilst in 1993 the grass block had significantly higher NH_4^+-N concentrations than the grass-clover block throughout July ($p<0.05$), and on 25 August ($p<0.05$) and 21 September ($p<0.01$). This was primarily due to the higher NH_4^+-N concentrations in the respective fallow and resown 1993 treatments.

Temporal patterns of NO_3^--N were smoother and revealed treatment differences more clearly than NH_4^+-N. Nitrate-N was the predominant form of mineral N in the disturbed treatments following ploughing and there were distinct differences between mineral N patterns in the two field seasons (Figures 4.1.3.2 and 4.1.3.3).

On 18 June 1992 there was no significant difference in soil mineral N concentrations in the undisturbed grass and grass-clover swards (Figure 4.1.3.3d), although NO_3^--N concentrations were significantly higher in the latter ($p<0.05$).

Following the initial rotavation on 22 June, NH_4^+-N concentrations showed a brief, but marked increase in the disturbed treatments (Figures 4.1.3.1a and b). Nitrate-N concentrations also increased sharply in the disturbed grass treatment and were significantly higher than in the undisturbed grass treatment on 14 July ($p\leq0.01$) (Figure 4.1.3.2).

During July 1992 NO_3^--N concentrations in the undisturbed grass-clover treatment were higher than during the rest of the trial. Nitrate-N concentrations in the undisturbed grass treatment remained low[2].

Following the second rotavation on 20 July, mineral N concentrations, predominantly as NO_3^--N, increased again in all the disturbed treatments. On the grass block this increase was sustained until 28 July whilst on the grass-clover block the increase was sustained further. On 28 July and 7 August mineral N concentrations in the undisturbed treatments were significantly lower than in the resown 1992 and fallow treatments ($p<0.05$).

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[1]In Figures 4.1.3.1, 4.1.3.2 and 4.1.3.3 where error bars are not shown, the errors from the nearest sampling dates in the same treatment are representative. Error bars are only given for the grass or grass-clover treatment in each figure but these errors are representative of the treatments in both sward blocks.

[2]On 14 July 1992 one anomalous value (265 kg NO_3^--N ha^{-1}) from plot CG 1 was removed.

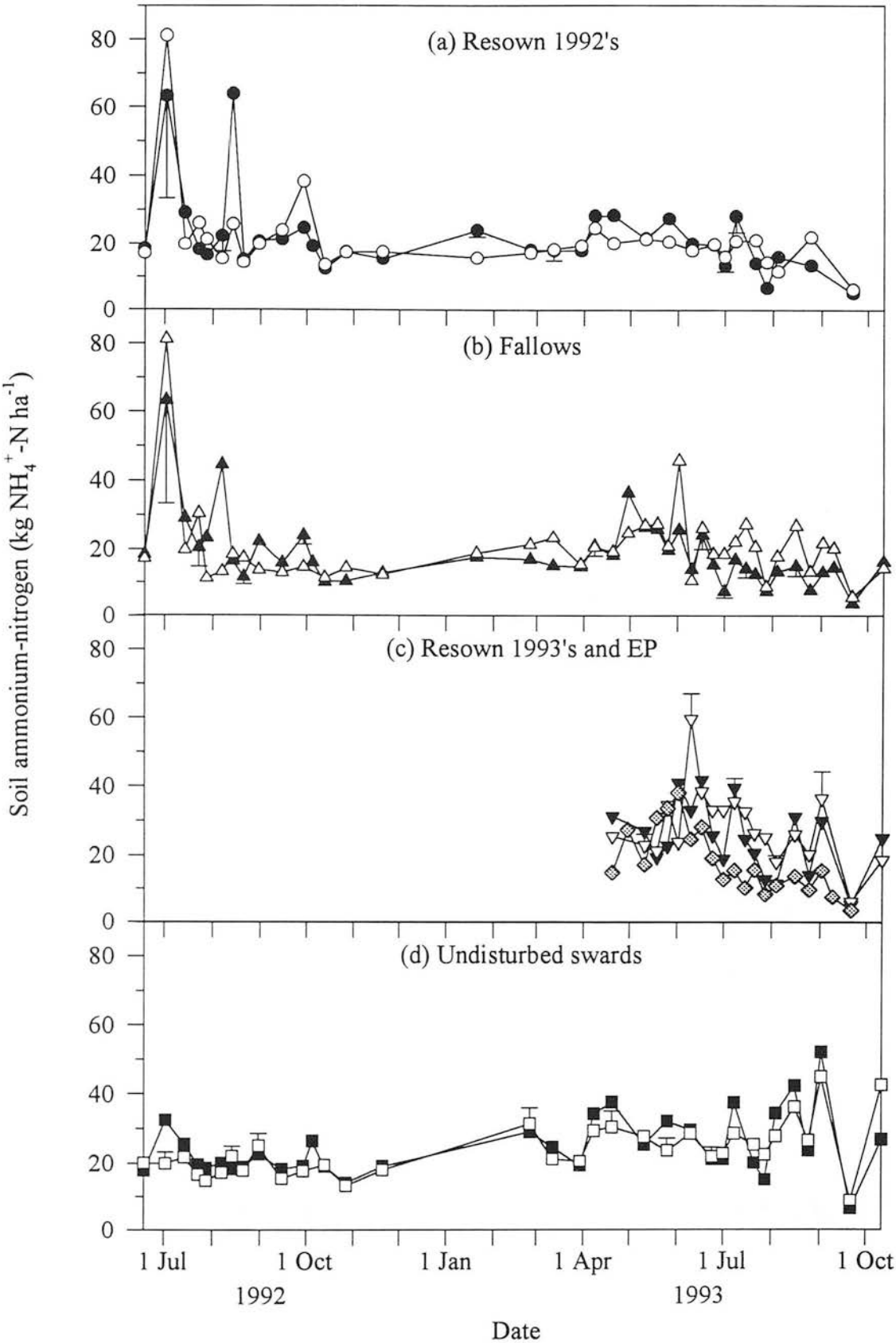

Figure 4.1.3.1 Soil ammonium-nitrogen contents, 0-20 cm depth (treatment means) on the grass clover (solid symbols, except the EP plots shaded symbol) and grass (open symbols) blocks of the Beechgrove field trial, 1992-3 (Bars = SE, d.f.=2).

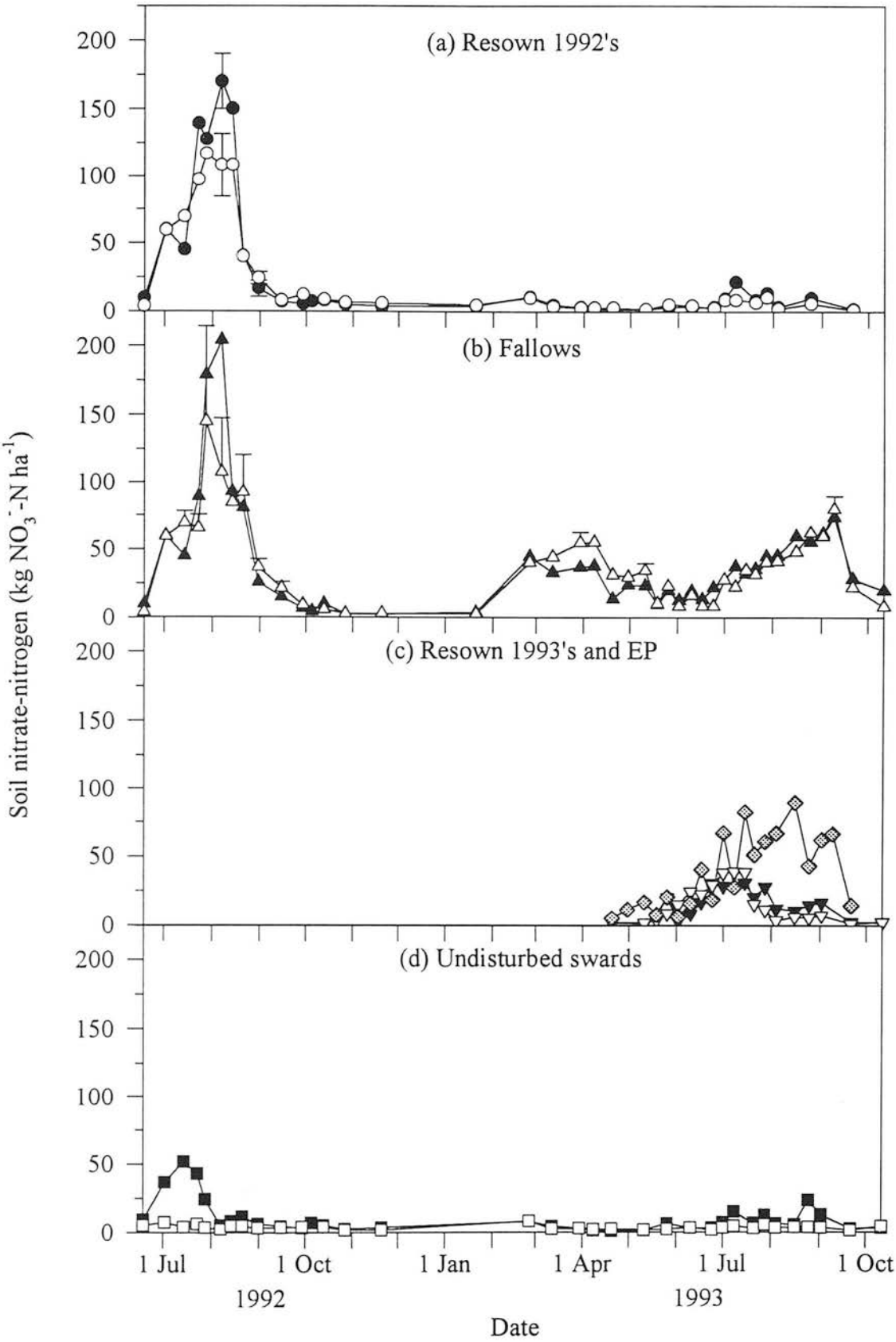

Figure 4.1.3.2 Soil nitrate-nitrogen contents, 0-20 cm depth (treatment means) on the grass clover (solid symbols, except the EP plots shaded symbol) and grass (open symbols) blocks of the Beechgrove field trial, 1992-3 (Bars = SE, d.f.=2).

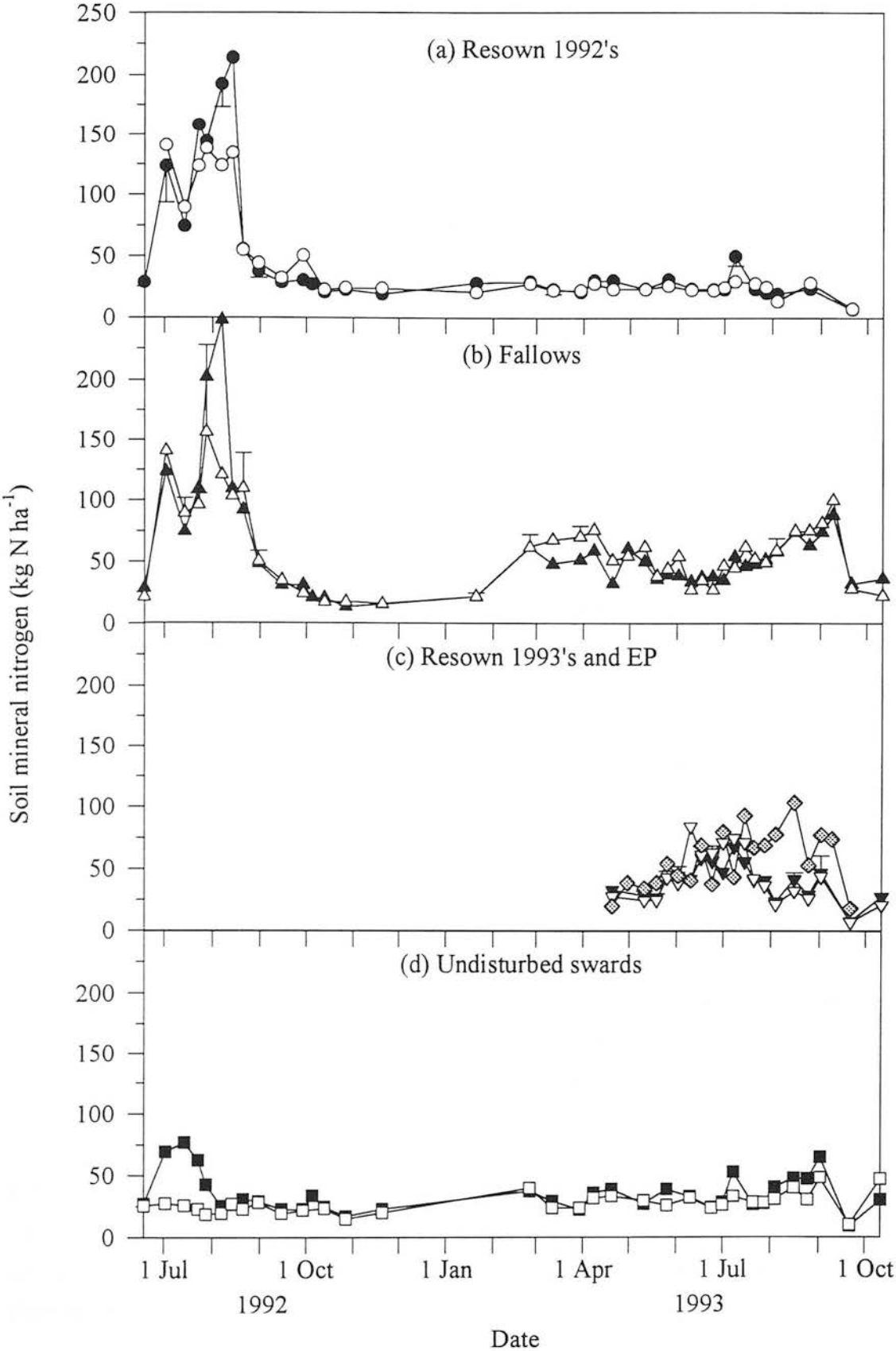

Figure 4.1.3.3 Soil mineral nitrogen contents, 0-20 cm depth (treatment means) on the grass clover (solid symbols, except the EP plots shaded symbol) and grass (open symbols) blocks of the Beechgrove field trial, 1992-3 (Bars = SE, d.f.=2).

Mineral N concentrations in the fallow treatments remained significantly higher than in the undisturbed ($p<0.001$) and resown 1992 treatments ($p<0.05$) on 14 and 21 August, respectively. Nitrate-N concentrations in the fallow treatments remained significantly higher than in the resown 1992 treatments on 31 August and 15 September ($p\leq0.05$). Mineral N concentrations in the undisturbed treatments were significantly lower than in the disturbed treatments on almost all subsequent sampling dates in 1992.

Between 12 March and 8 April mineral N concentrations in the fallow treatments were significantly higher than in the resown 1992 and undisturbed treatments ($p<0.001$). On 20 April mineral N concentrations in the PGF treatment were significantly higher than in the PGCF treatment ($p<0.05$).

On 25 February and 20 April NH_4^+-N concentrations in the undisturbed treatments were significantly higher than in the resown 1992 treatments ($p<0.05$). Following this "spring bite" plateau, NH_4^+-N concentrations in all treatments undisturbed in 1993, except the grass fallow, generally fell until 1 July and treatment differences remained similar.

Following the ploughing out of the EP plot on 5 April, t-tests showed NH_4^+-N concentrations were significantly lower than in all other treatments on 20 April ($p<0.05$). Subsequently, NH_4^+-N concentrations increased until 2 June when they were significantly higher than in the fallow, resown 1992 and 1993 treatments on the grass-clover block ($p<0.05$). Nitrate-N concentrations in the EP plot also increased steadily until 11 May. Consequently, mineral N concentrations in the EP plot were significantly higher than in the resown 1992 treatment on 11 May ($p<0.05$). Mineral N concentrations in the fallow treatments also increased between 20 April and 11 May, when they were significantly higher than in all other treatments ($p<0.001$), except the EP plot ($0.05<p<0.1$).

Mineral N concentrations in the resown 1993 treatments showed no immediate response to ploughing (11 May). Nitrate-N concentrations in the resown 1993 treatments did increase slightly but remained significantly lower than in the fallow treatments on 19 May ($p<0.05$).

On 26 May mineral N concentrations in the resown 1993 and fallow treatments were significantly higher than in the resown 1992 treatments ($p<0.05$). On 26 May mineral

N concentrations in the EP plot were higher than in all other treatments in the grass-clover block ($0.05<p<0.1$).

Following rotavation (26 May), NH_4^+-N concentrations in the resown 1993 treatments were significantly higher than in all other treatments on 10 June ($p<0.05$). On 10 June mineral N in the resown grass 1993 treatment was significantly higher than in the respective grass-clover treatment ($p<0.001$). In the EP plot a fluctuating upward trend was soon established after rotavation. On 15 July, NO_3^--N concentrations in the EP plot were significantly higher than in the fallow and resown 1993 treatments on the grass-clover block ($p<0.01$).

In a reversal of the pattern in 1992, NO_3^--N concentrations in the resown grass 1993 treatment peaked later, and higher than in the respective grass-clover treatment. Between 10 June and 8 July mineral N concentrations in the resown 1993 treatments were significantly higher than in the undisturbed ($p<0.01$), resown 1992 ($p<0.01$) and fallow treatments ($p<0.05$).

After 10 June, NH_4^+-N concentrations generally fell until 28 July. Ammonium-N concentrations in the fallow treatments continued to be significantly lower than in the undisturbed treatments until sampling ceased ($p<0.05$) and were frequently significantly lower than in the resown 1993 treatments.

Despite increasing slightly after 11 May, NO_3^--N concentrations in the resown 1992 and undisturbed treatments were significantly lower than in the fallow treatments from 11 May until sampling ceased ($p<0.05$).

Nitrate-N concentrations in the resown 1993 treatments were only significantly higher than in the fallow treatments on 17 and 24 June ($p<0.05$) and from 21 July onwards were significantly lower than in the fallow treatments ($p<0.01$). Mineral N concentrations in the resown 1993 and fallow treatments remained significantly higher than in the undisturbed ($p<0.05$) and resown 1992 ($p<0.01$) treatments until 28 July. However, by 4 August mineral N concentrations in the resown 1993 treatments were significantly lower than in the undisturbed treatments ($p<0.05$).

After 28 July several treatments showed a sustained, upward trend in mineral N concentrations. Between 15 July and 16 August mineral N concentrations in the EP plot were significantly higher than in the undisturbed ($p<0.01$), resown 1992

($p<0.01$) and resown 1993 treatments ($p<0.05$) on the grass-clover block. Between 4 August and 2 September mineral N concentrations in the fallow treatments were significantly higher than in all other treatments ($p<0.05$).

All treatments showed a sharp decrease in mineral N to 21 September. Despite this, mineral N concentrations in the fallow treatments were still significantly higher than in all other treatments ($p<0.001$). Mineral N concentration in the EP plot remained significantly higher than all bar the fallow treatments ($p\leq0.01$).

4.1.4 Gaseous nitrogen losses in the Beechgrove field trial

In 1992, due to the unbalanced experimental design, a general linear model was used to assess the effects of sward type and treatment on nitrous oxide fluxes. On 3 and 10 July such analysis was not possible due to reduced replication and therefore on these dates one-way analysis of variance and t-tests were used. In 1993, two-way analysis of variance was used on most occasions.

On 2 and 3 July 1992 fluxes from the grass-clover fallow treatment were significantly higher than those from the undisturbed grass-clover treatment ($p<0.01$ and $p<0.05$, respectively). This treatment effect was also apparent in the grass block ($p<0.05$). Fluxes from the undisturbed grass sward were significantly lower than from the grass-clover sward ($p<0.01$), but the effect of sward type was not quite significant in the fallow treatments ($0.05<p<0.1$).

On 6 July fluxes from the fallow treatments were not quite significantly higher than those from the undisturbed swards ($0.05<p<0.1$). Despite the apparently higher emissions from the grass-clover block, there was no significant effect of sward type. However, analysing log-transformed data, Bergos (1992) found that between 3 and 13 July there was a significant effect of sward type ($p<0.01$).

After 6 July, fluxes fell substantially although fluxes from the grass-clover and grass fallow treatments remained significantly higher than those from the undisturbed grass sward ($p\leq0.001$ and $p<0.05$, respectively). Fluxes from the fallow treatments remained significantly higher than those from the undisturbed swards on 15 ($p<0.05$), 23 ($p<0.001$) and 29 July ($p<0.01$).

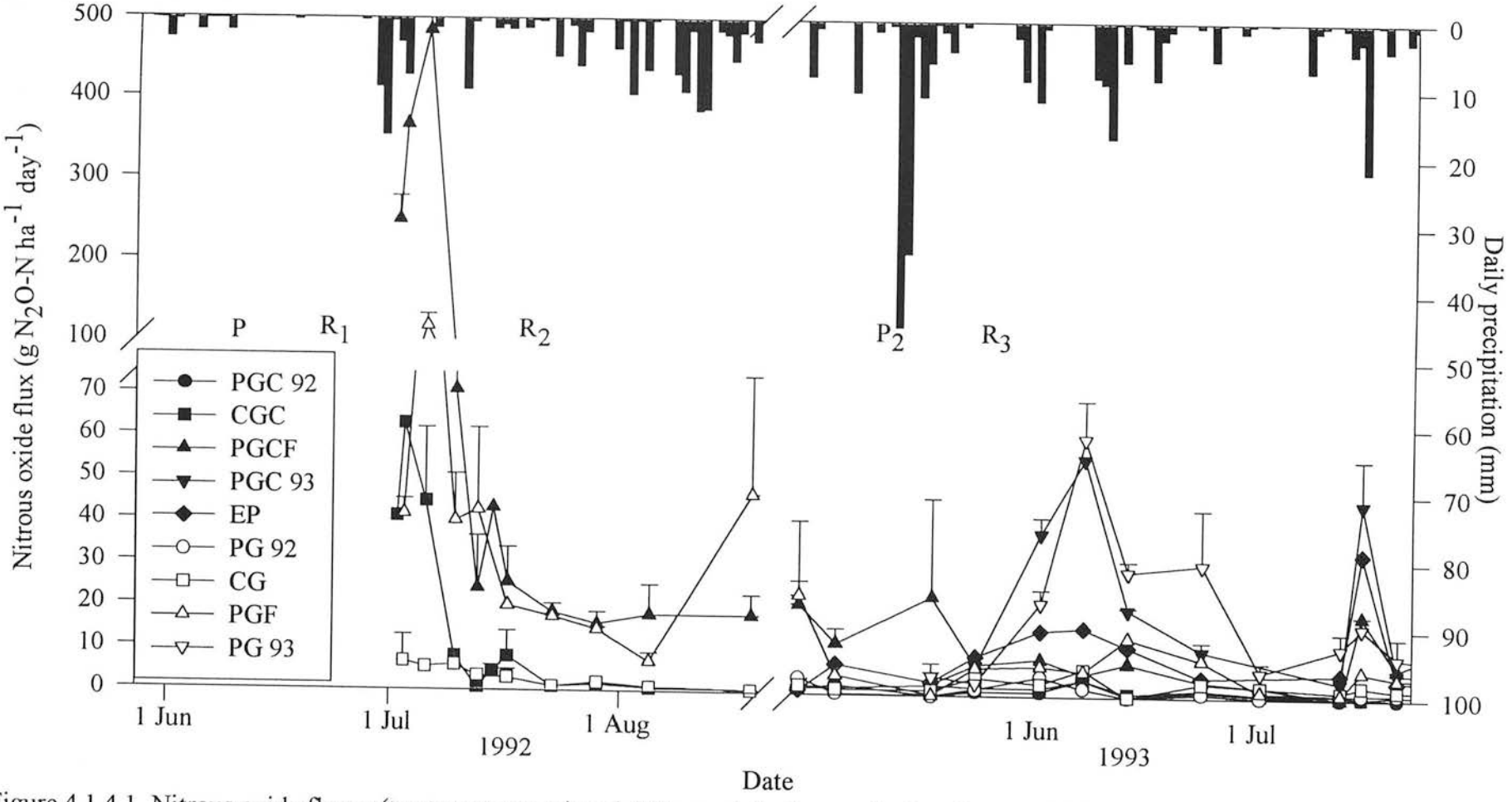

Figure 4.1.4.1 Nitrous oxide fluxes (treatment means) and daily precipitation on the Beechgrove field trial, 1992-3 (Bars = SE)[a]. (P_1,R_1 and R_2=ploughing, first and second rotavation,respectively, of the resown 1992 and fallow treatments. P_2 and R_3=ploughing and rotavation, respectively, of the resown 1993 treatments.

[a] Where error bars are not shown, bars from the nearest sampling dates in the same treatment are representative. In 1993, errors are only shown for the resown 1993 and fallow treatments. D.f.=1in 1992, 2 in 1993.

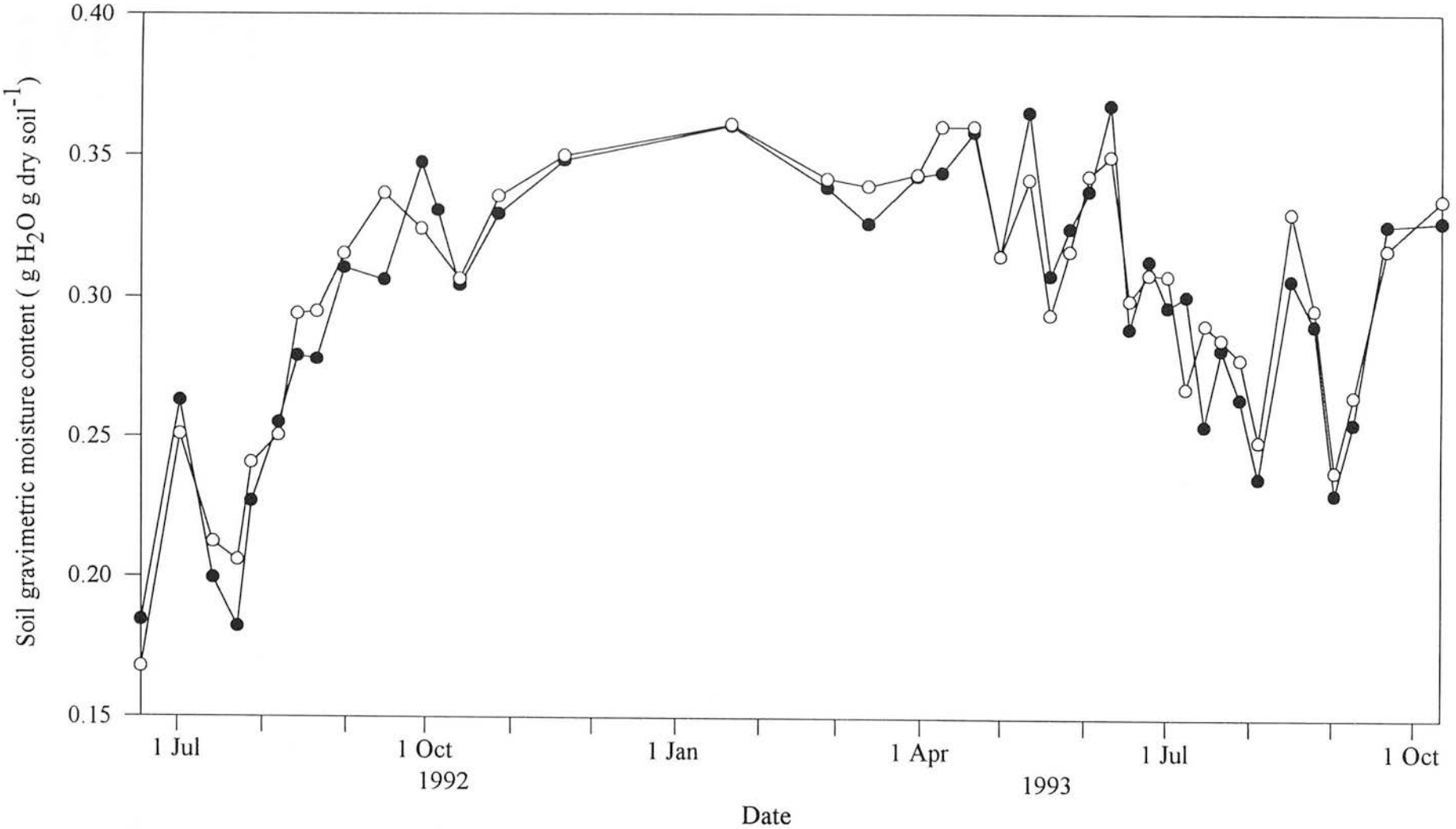

Figure 4.1.4.2 Mean soil gravimetric moisture content (0-20cm) for the grass-clover (solid symbols) and grass (open symbols) blocks on the Beechgrove field trial, 1992-3 (n=3).

On 19 August fluxes from the fallow treatments were once again significantly higher than those from the undisturbed swards ($p<0.05$). The cumulative nitrous oxide and denitrification fluxes for the sampled period are given in Table 4.1.4.

In 1993, there were two main periods of flux activity (Figure 4.1.4.1). As in 1992, fluxes were generally higher from those treatments with recently incorporated sward residues or those left fallow. In 1993 emissions were much lower than in 1992.

On 30 April, fluxes from the fallow treatments were higher than from the undisturbed and resown 1992 treatments (Figure 4.1.4.1). The EP plot had the lowest flux. Treatments ploughed in 1992 showed higher fluxes on 30 April than during the rest of the 1993 season.

In contrast to the other treatments, between 30 April and 5 May fluxes from the EP plot increased and were significantly higher than from the resown 1992 and undisturbed treatments in the grass-clover block ($p<0.05$). Fluxes from the fallow treatments were still significantly greater than from the resown 1992 and undisturbed treatments ($p<0.05$). On 24 May the EP plot had the highest fluxes, significantly greater than the resown 1992 ($p\leq0.001$) and undisturbed treatments ($p<0.01$).

On 2 June fluxes from the resown 1993 treatments were significantly higher than those from all other replicated treatments ($p<0.001$). T-tests revealed that the flux from the EP plot was significantly higher than from the resown 1992 and undisturbed treatments ($p<0.01$). Fluxes from the resown grass-clover 1993 treatments were significantly higher than from the EP plot on 2 June ($p<0.05$). Fluxes from the resown grass 1993 treatment were significantly lower than from the respective grass-clover treatment ($p<0.05$).

On 8 June emissions from the resown 1993 treatments and EP plot remained significantly higher than all other treatments, $p<0.001$ and $p<0.05$, respectively. Fluxes from the resown 1993 treatments were significantly higher than from the EP plot ($p<0.05$).

On 24 June fluxes from the resown 1993 treatments remained significantly higher than from the undisturbed, resown 1992 ($p<0.01$) and fallow treatments ($p<0.05$). Fluxes from the undisturbed and resown 1992 treatments were also significantly lower than from the EP plot ($p<0.01$). Fluxes from the resown grass 1993 treatment

were significantly higher than from the respective grass-clover treatment on 14 June ($p<0.05$).

On 2 July fluxes from all treatments were below 6 g N_2O-N ha^{-1} day^{-1}. On 13 July fluxes remained small, with the exception of the resown grass 1993 treatment. Similar flux increases may have occurred from the resown grass-clover 1993 treatment and EP plot, but no data were available.

Between 13 and 16 July large flux increases were observed from those treatments ploughed out in 1993 and the fallow treatments, and previous treatment differences re-emerged. On 16 July fluxes from the grass-clover block were significantly higher than those from the grass block ($p<0.05$). Fluxes from the resown 1993 treatments were significantly higher than from the undisturbed, resown 1992 ($p<0.001$) and fallow treatments ($p<0.05$). The flux from the EP plot was significantly higher than from the resown grass 1993 ($p<0.05$), undisturbed and resown 1992 treatments ($p<0.001$).

This second period of increased emissions was less prolonged than the first and, by 21 July, all fluxes were below 10 g N_2O-N ha^{-1} day^{-1}. Fluxes from the EP plot and grass fallow treatment were significantly higher than from the undisturbed and resown 1992 treatments ($p<0.05$).

On 21 September emissions from the grass-clover block were higher than from the grass block ($0.05<p<0.1$). Fluxes from the EP plot were significantly greater than from the resown 1993 and undisturbed treatments ($p\leq0.001$). Subsequently fluxes from all treatments showed a downward trend.

The total N_2O fluxes between 30 April and 21 July are shown in Table 4.1.4. Fluxes were not integrated beyond this date because sampling was considered to be too infrequent for fluxes to be interpolated between sample dates.

Table 4.1.4 Gaseous nitrogen fluxes (treatment means) from the Beechgrove trial in 1992[a] and 1993[b] (Standard errors in parentheses, d.f.=1 in 1992, 2 in 1993).

	Nitrous Oxide Flux (kg N_2O-N ha^{-1})		Denitrification flux (kg N_2-N + N_2O-N ha^{-1})	
Treatment	1992	1993	1992[c]	1993
PGC92		0.1 (0.0)		0.3 (0.0)[d]
PG92		0.1 (0.0)		0.4 (0.1)[d]
CGC	0.5 (0.1)	0.1 (0.0)	1.6 (0.4)	0.5 (0.1)[d]
CG	0.1 (0.0)	0.2 (0.0)	0.4 (0.1)	0.8 (0.0)[d]
PGCF	3.7 (0.5)	0.8 (0.1)	17.5 (2.8)	4.1 (0.8)[e]
PGF	1.5 (0.0)	0.5 (0.3)	7.8 (0.0)	2.7 (1.6)[e]
EP		0.7 (0.1)		4.0 (0.5)[e]
PGC93[f]		1.2 (0.1)		6.7 (0.6)[e]
PG93[f]		1.3 (0.2)		7.0 (0.9)[e]

[a] Fluxes from the grass block are calculated between 3 July and 19 August, fluxes from the grass clover block are calculated between 2 July and 19 August.
[b] Fluxes are calculated between 30 April and 21 July.
[c] Calculated using measured N_2O+N_2:N_2O ratios where available, and an average of the measured ratios for the other dates.
[d] Assuming an N_2+N_2O:N_2O ratio of 3.7, the measured ratio for the CGC treatment in 1992.
[e] Assuming an N_2+N_2O:N_2O ratio of 5.4, the average measured ratio from the ungrazed treatment in the laboratory experiment (see Table 4.4.1.6).
[f] Assumes fluxes are the same as from the relevant undisturbed treatments prior to ploughing on 11 May.

4.1.5 Leaching studies on the Beechgrove field trial

4.1.5.1 Soil water nitrogen concentrations in porous ceramic cups

Due to incomplete cup installation, prior to 27 August the only possible treatment comparisons were one-way analysis of variance between the individual plots which had three cups installed (PGCF 3, PGC92 2 and CGC 1). After 27 August, 28 September and 5 October t-tests were also possible to compare these plot means with the fallow, resown 1992 and undisturbed treatment means on the grass block, respectively. By 16 October all cups were installed, and complete statistical analysis was possible.

On the first sampling date, soil water NO_3^--N concentrations in plot PGCF 3 were significantly higher than in plots PGC92 2 and CGC 1 ($p<0.05$). Concentrations in plot PGC92 2 were not quite significantly higher than in plot CGC 1 ($0.05<p<0.1$) (Figure 4.1.5.1.1). On 21 August, soil water NO_3^--N concentrations in plot PGCF 3 were not quite significantly higher than in plot PGC92 2 ($0.05<p<0.1$) whilst, due to one cup, the mean concentration in plot CGC 1 was 40 μg NO_3^--N ml^{-1} (section 5.5). After 21 August, porous cups produced samples more consistently.

Between 21 and 31 August soil water NO_3^--N concentrations in plot PGCF 3 remained significantly higher than in plot PGC92 2 ($p<0.05$) and the grass fallow treatment ($p<0.05$). After 31 August soil water NO_3^--N concentrations fell sharply in plot CGC 1 and, by 21 September, were below 0.5 μg NO_3^--N ml^{-1}. Concentrations remained this low until sampling ceased in 1993. After 7 September soil water NO_3^--N concentrations also started to fall in plot PGC92 2, and were no longer higher than in plot CGC 1 ($P<0.05$).

Between 7 and 21 September soil water NO_3^--N concentrations in plot PGCF 3 were significantly higher than in plots PGC92 2 ($p<0.05$) and CGC 1 ($p<0.01$). On 14 September soil water NO_3^--N concentrations in the grass fallow treatment were significantly higher than in plot CGC 1 ($p<0.05$), though not quite significantly higher than in plot PGC92 2 ($0.05<p<0.1$).

Between 21 September and 21 October, concentrations in PGCF 3 remained significantly higher than in plots CGC 1 ($p<0.001$) and PGC92 2 ($p<0.01$), and the resown 1992 and undisturbed treatments on the grass block ($p<0.01$).

Soil water NO_3^--N concentrations in the grass fallow treatment peaked at 135 μg NO_3^--N ml^{-1} on 16 October, significantly higher than all treatments except the grass-clover fallow treatment ($p<0.05$).

Soil water NO_3^--N concentrations in the grass block were significantly higher than in the grass-clover block between 16 October and 18 November ($p<0.05$).

Soil water NO_3^--N concentrations in the fallow treatments remained significantly higher than those in the resown 1992 and undisturbed treatments ($p<0.05$) until 2 December. Between 7 January and 10 February, soil water NO_3^--N concentrations in

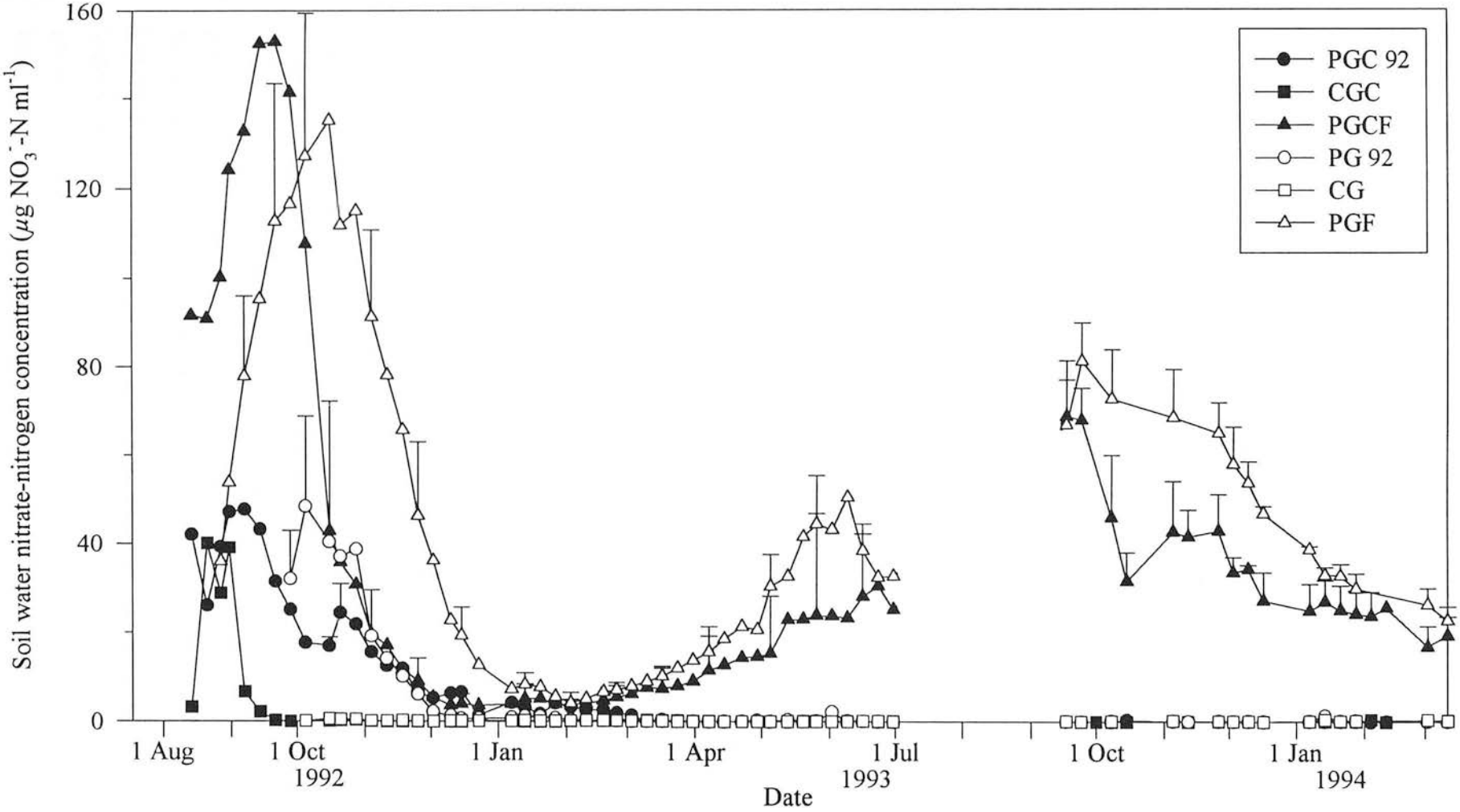

Figure 4.1.5.1.1 Nitrate-nitrogen concentrations (treatment means) in porous cup samples on the Beechgrove field trial, 1992-4 (Bars = SE)[a].
[a] D.f.=2. Where error bars are not shown, bars from the nearest sampling dates in the same treatment are representative. After 21 October 1992 errors are only given for the fallow treatments.

the fallow treatments were frequently significantly higher than the resown 1992 and undisturbed treatments, particularly in the grass block.

After 20 January there was a slight, but steady rise in soil water NO_3^--N concentrations in the undisturbed treatments until 24 February. In contrast, soil water NO_3^--N concentrations in the resown 1992 treatments fell during this period. By 31 March concentrations in both of these treatments were below 0.1 μg NO_3^--N ml^{-1}.

Soil water NO_3^--N concentrations in the fallow treatments were significantly higher than in the resown 1992 and undisturbed treatments on 24 February, 3 and 24 March ($p<0.05$). The fallow treatments had significantly higher soil water NO_3^--N concentrations than the undisturbed and resown 1992 treatments on all dates when two-way analysis of variance was possible ($p<0.05$).

After 30 June cups failed to yield samples, in agreement with the cessation of drainage at Glencorse (Figure 4.1.5.1.2). Between 30 July and 3 September there were small volumes of drainage at Glencorse but cups at Beechgrove failed to yield samples when tested on 27 August. The first successful cup sampling was on 17 September coinciding with the first major drainage at Glencorse (58 mm drainflow between 13 and 17 September). Thereafter porous cups yielded samples more consistently and dates on which cups did not yield samples coincided with periods of low drainflow.

Throughout the 1993-4 cup sampling period, soil water NO_3^--N concentrations in the resown 1993 (not shown), resown 1992 and undisturbed treatments were below 1 μg NO_3^--N ml^{-1}, and often below the reliable detection limit of 0.25 μg NO_3^--N ml^{-1}, showing no clear temporal patterns (Figure 4.1.5.1.1). Two-way analysis of variance, where possible, showed that soil water NO_3^--N concentrations in the fallow treatments were significantly higher than in all other treatments ($p<0.001$).

Where only one-way analysis was possible, soil water NO_3^--N concentrations in the grass fallow treatment were always significantly higher than in all other treatments, as was the grass-clover treatment on all bar four occasions ($p<0.05$). Between 3 December 1993 and 7 January 1994 two-way analysis of variance found that soil water NO_3^--N concentrations were significantly higher in the grass block than the grass-clover block ($p<0.05$), primarily due to concentrations in the grass fallow treatment being higher than in the grass-clover fallow treatment.

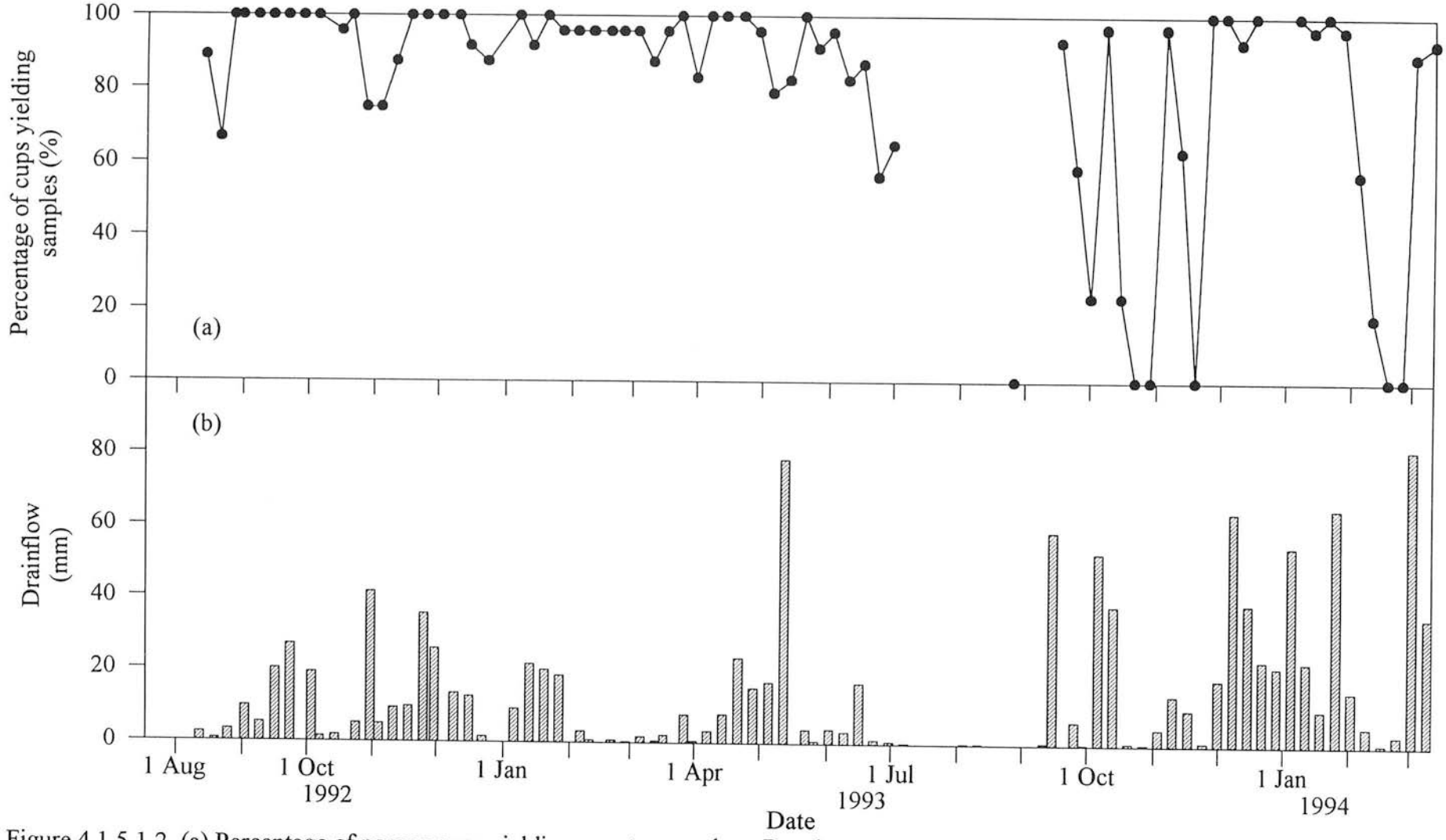

Figure 4.1.5.1.2 (a) Percentage of porous cups yielding a water sample at Beechgrove, 1992-4. (b) Drainflow as measured from the hydrologically isolated plots at Glencorse.

4.1.5.2 Estimated nitrate-N leaching loads from the Beechgrove field trial

In the 1992-3 drainage season, estimated leaching loads from the resown 1992 treatments were significantly higher than from the undisturbed grass treatment ($p<0.05$) though not quite significantly higher than the undisturbed grass-clover treatment ($0.05<p<0.1$). Leaching losses from the fallow treatments were significantly higher than from the resown 1992 ($p<0.05$) and undisturbed treatments ($p<0.01$). There was no significant effect of sward type on leaching losses.

In the 1993-4 drainage season leaching losses from the fallow treatments were significantly higher than from all other treatments ($p<0.001$). In the grass-clover block, leaching losses from the undisturbed sward were not quite significantly higher than from the resown 1992 treatment ($0.05<p<0.1$). Leaching losses from the grass block were significantly higher than from the grass-clover block ($p<0.01$), primarily due to the greater losses from the grass fallow treatment ($p<0.05$).

Table 4.1.5.2 Estimated nitrate-N leaching loads (treatment means) from the Beechgrove field trial (Standard errors in parentheses, d.f.=2).

	Nitrate-N leaching load (kg NO_3^--N ha^{-1})	
Treatment	1992-3[a]	1993-4[b]
CGC	3.9 (1.8)	1.2 (0.2)
PGC92	49.9 (18.5)	0.5 (0.2)
PGCF	106.8 (77.2)	197.4 (21.2)
PGC93	-	1.7 (0.7)
CG	0.7 (0.1)	1.3 (0.6)
PG92	44.3 (15.1)	1.3 (0.5)
PGF	250.6 (45.3)	296.3 (20.1)
PG93	-	1.0 (0.4)

[a] Calculated for the period 14 August 1992-30 June 1993.
[b] Calculated for the period 17 September 1993-11 March 1994.

4.1.5.3 Bromide tracer experiment

Following Br tracer application, cup 1 showed an unexpected and anomalous peak (section 5.6) whilst Br concentrations in all other cups remained fairly stable until 24 March (Figure 4.1.5.3). After 24 March, Br concentrations in all cups increased and, by 14 April, were significantly higher than in the control sample at both 40 cm ($p<0.01$) and 55 cm depths ($p<0.05$). (Concentrations in the control sample remained significantly lower than those in cup samples from 40 cm and 55 cm depths ($p<0.05$) on all bar one occasion until cup sampling ceased.) Bromide concentrations peaked on 5 May, those at 40 cm depth being significantly higher than those at 55 cm on 28 April and 5 May ($p<0.05$). Subsequently, Br concentrations in cup samples fell until 9 June, the fall being greater at 40 cm depth. Concentrations increased slightly until 30 June, after which cups no longer yielded samples during July and August.

When sampling resumed, Br concentrations in cup samples from 40 cm depth showed a general downward trend until 14 January 1994 and were then stable. At the 55 cm depth Br concentrations were more erratic, showing a slight downward trend. Bromide concentrations at 40 cm depth were only significantly higher than at 55 cm depth on 8 October ($p<0.05$).

The calculated cumulative bromide loads at the two depths are shown in Table 4.1.5.3. There were no significant differences in the bromide loads at the two depths. Estimated vegetation uptake of Br was 2.3 g Br m^{-2} (Appendix 1.1).

Table 4.1.5.3. Calculated cumulative Br loads[a] on 30 June 1993, 8 October 1993 and 11 March 1994 after bromide tracer application (5 g Br m^{-2}) (Standard errors in parentheses, d.f.=2 at 40 cm, 1 at 55 cm).

	Cumulative Br leaching load (g Br m^{-2})	
Date	Cup placement depth	
	40 cm	55 cm
30 June 1993	0.38 (0.04)	0.27 (0.04)
8 October 1993	0.68 (0.08)	0.45 (0.04)
11 March 1994	1.33 (0.13)	1.11 (0.03)

[a] Calculated bromide load from control samples deducted.

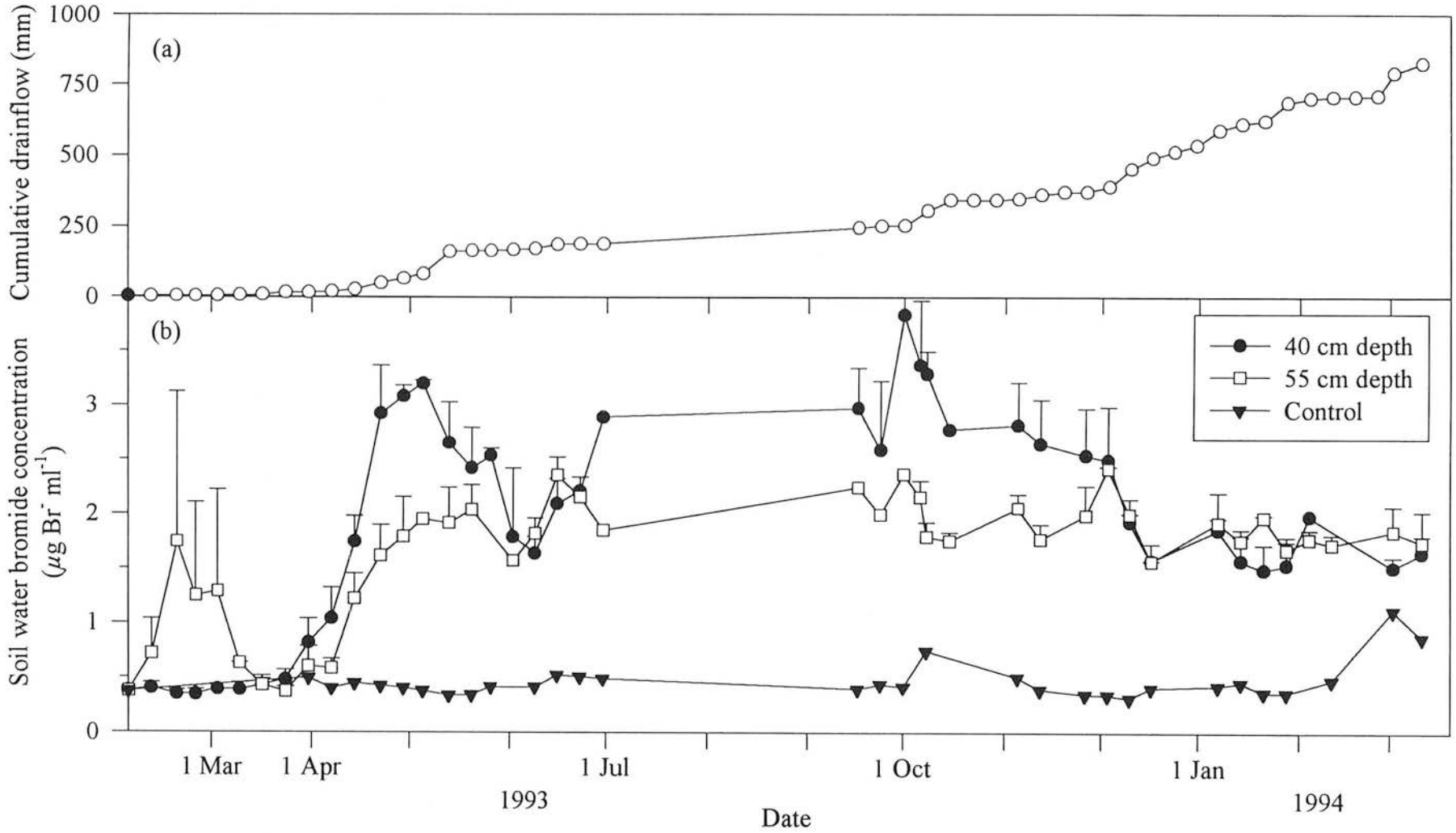

Figure 4.1.5.3 (a) Cumulative drainflow as measured from the hydrologically isolated plots at Glencorse. (b) Mean bromide concentrations in porous cup samples on the Beechgrove field trial, 1993-4 (Bars = SE, d.f.=2 at 40cm, 1 at 55cm).

4.1.5.4 Comparison of soil water nitrate-N concentration measured using deep soil cores and porous cup sampling

In October 1992, NO_3^--N concentrations in cup samples and soil cores were qualitatively similar in the fallow treatments, ranking plots within treatments in the same order and with the same magnitude of difference between them (Figure 4.1.5.4.1 and 4.1.5.4.2). This qualitative similarity between the sampling techniques appeared to exist prior to and after the core sampling date (data not shown). However, NO_3^--N concentrations in cup samples tended to be higher and in the grass fallow treatment were approximately double those in soil cores (Figure 4.1.5.4.2).

In the resown 1992 treatments there was less agreement between the two sampling techniques. In the resown grass 1992 treatment, cup and core samples matched well in plots 1 and 3, though again NO_3^--N concentrations in the cup samples were slightly higher (Figure 4.1.5.4.3a). In plot 2 however, NO_3^--N concentrations in the cup sample were twice as high as those in the soil core. In the resown grass-clover 1992 treatment, NO_3^--N concentrations in cores showed distinct differences between plots which were not detected in the cup samples, and concentrations were never similar to the cores (Figure 4.1.5.4.3b).

In January 1993 NO_3^--N concentrations in cup samples were generally slightly lower than those measured in soil cores, although those plots which maintained higher concentrations in core samples also showed higher concentrations in cup samples.

In October 1993 the cup and core samples in the grass-clover fallow treatment only showed good agreement in plot 3, although treatment averages using the two techniques were very similar. In the grass fallow treatment, whilst cup and core samples ranked the plots in the same order, NO_3^--N concentrations in cup samples were roughly double those in core samples. In January 1994 cup and core samples continued to show this difference in the grass fallow treatment, whilst in the grass-clover fallow the two techniques compared quite favourably.

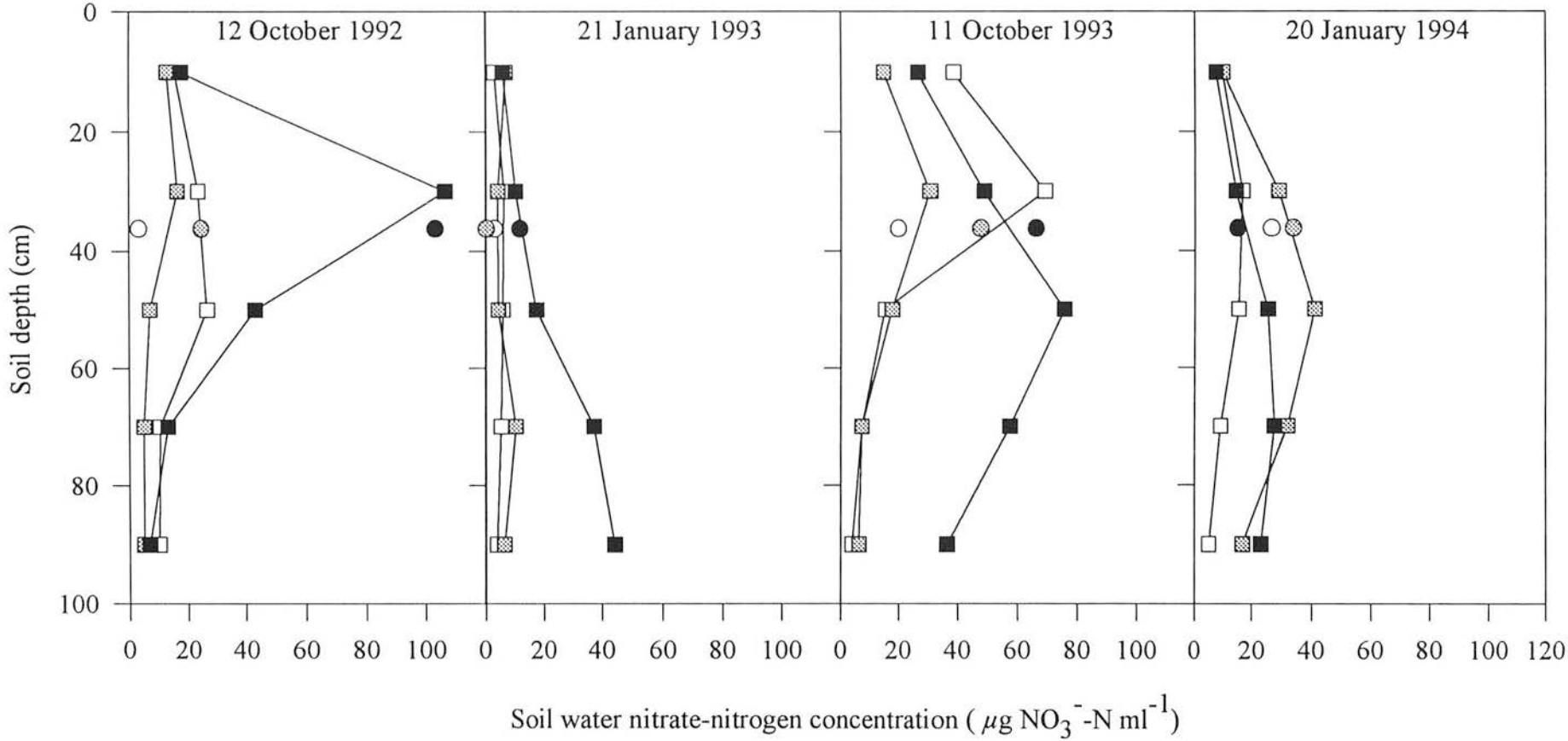

Figure 4.1.5.4.1 Nitrate-nitrogen concentrations in water samples from porous cups (round symbols) and soil cores (square symbols) on individual plots[a] of the ploughed grass-clover fallow treatment at the Beechgrove trial, 1992-4.

[a] Plot 1=open symbols, plot 2=shaded symbols, plot 3=solid symbols.

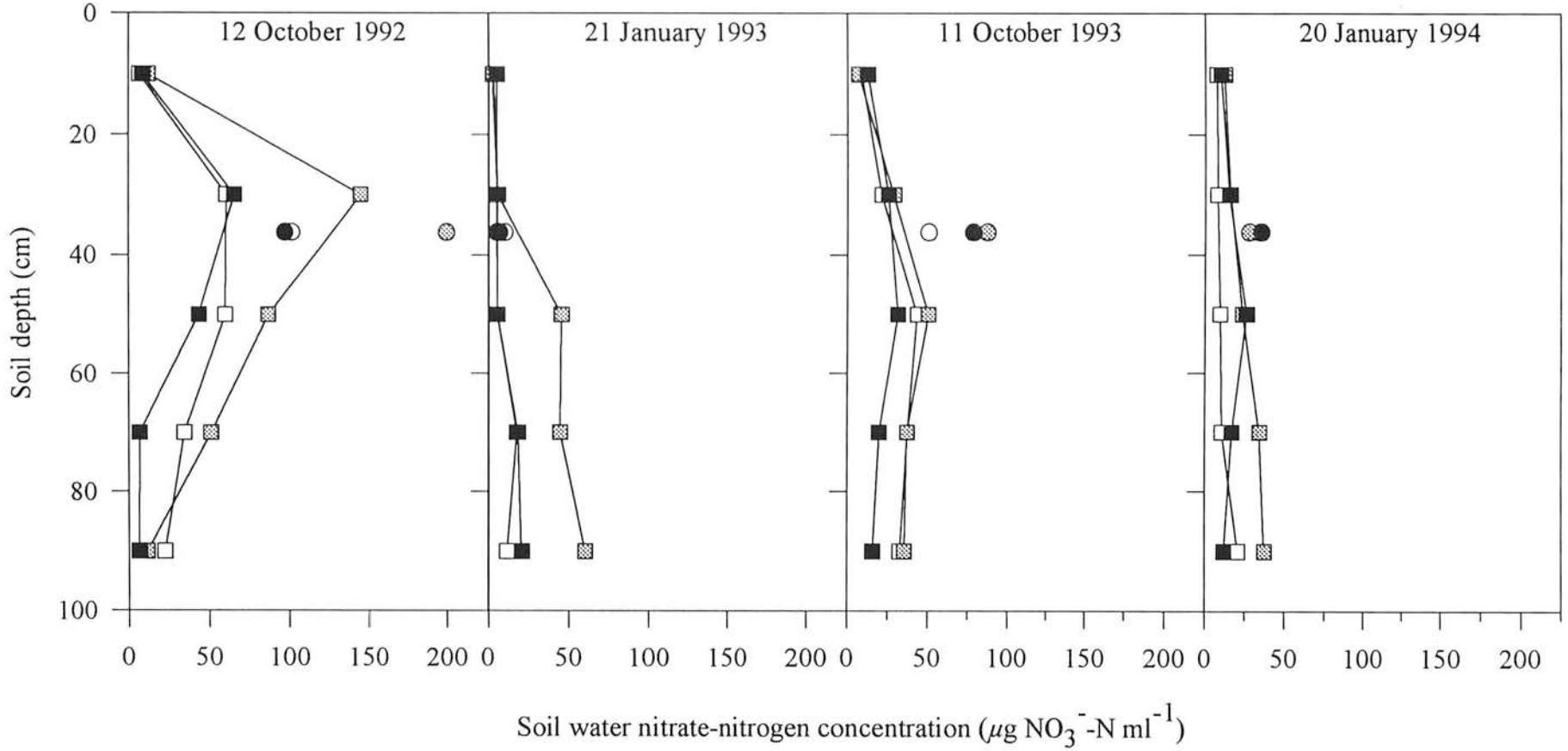

Figure 4.1.5.4.2 Nitrate-nitrogen concentrations in water samples from porous cups (round symbols) and soil cores (square symbols) on individual plots[a] of the ploughed grass fallow treatment at the Beechgrove trial, 1992-4.

[a] Plot 1=open symbols, plot 2=shaded symbols, plot 3=solid symbols.

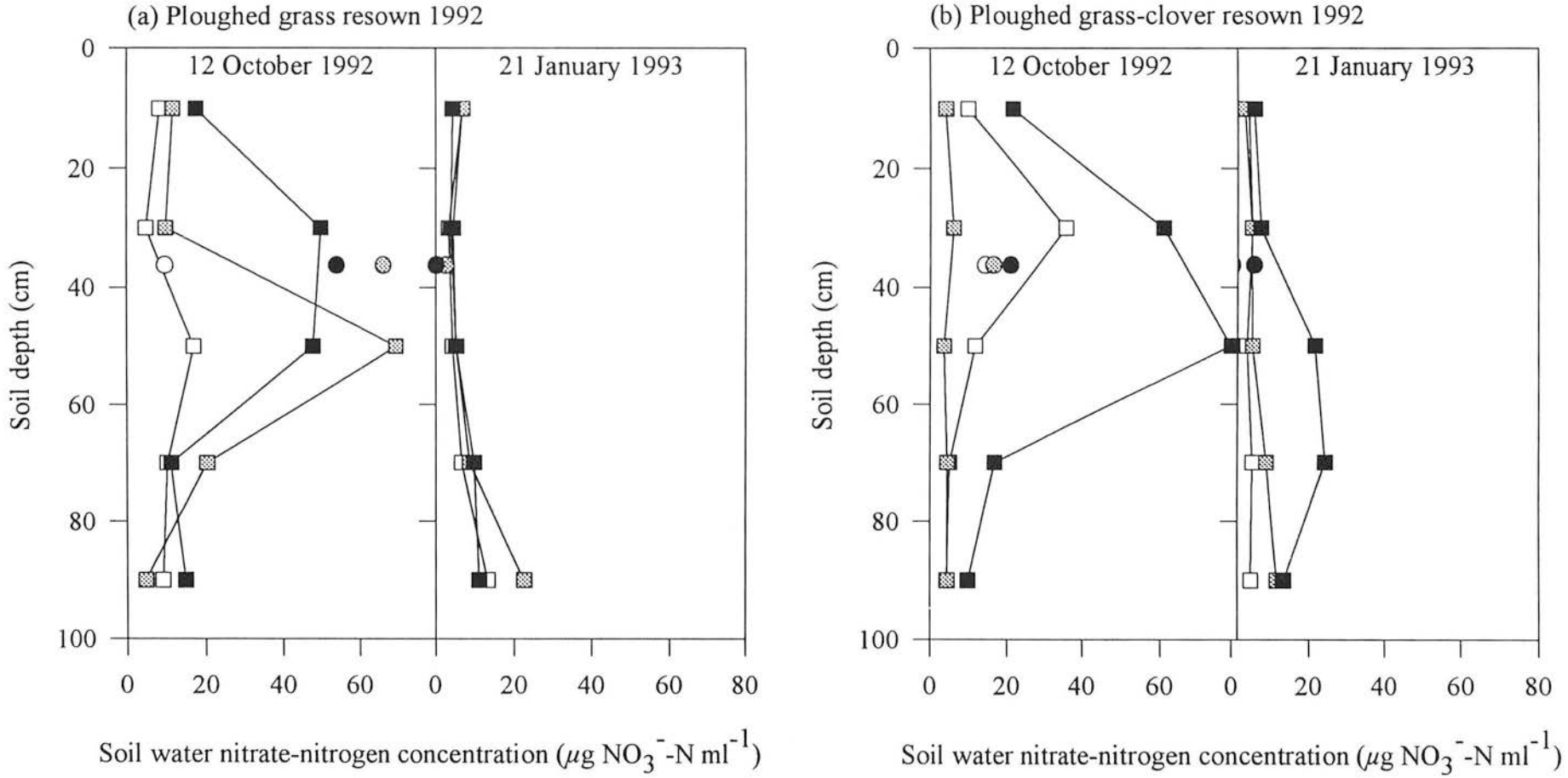

Figure 4.1.5.4.3 Nitrate-nitrogen concentrations in water samples from porous cups (round symbols) and soil cores (square symbols) on individual plots on the resown 1992 treatments at the Beechgrove trial, 1992-4.

[a] Plot 1=open symbols, plot 2=shaded symbols, plot 3=solid symbols.

4.1.6 Above-ground sward composition, plant nitrogen content, plant dry matter production and nitrogen uptake

The clover content of the two undisturbed swards showed very different patterns during the field trial (Table 4.1.6). The clover content of the grass-clover sward was not quite significantly higher than that of the grass sward on 11 June and 15 October ($0.05<p<0.1$), in agreement with earlier quadrat analysis of clover percentage ground cover ($p<0.001$). Swift *et al.* (1993) also observed higher clover contents on the grass-clover than the grass swards in 1991 and 1992. However, between 15 October 1992 and 4 May 1993, the clover content of the grass-clover sward fell whilst that of the grass sward increased. Consequently, the clover content of the grass sward was significantly higher on 4 May ($p\leq0.01$) and, by 6 June, was double that of the grass-clover sward.

Table 4.1.6 Sward clover contents and clover N contents during the Beechgrove field trial (Standard errors in parentheses[a]).

Sampling Date	Sward Clover content (% in DM)		Clover Nitrogen content (% N)	
Sward	Grass	Grass-clover	Grass	Grass-clover
Season means[b]				
1991	<4	17.6	No Data	No Data
1992	<7	18.2	No Data	No Data
Quadrat analysis				
27 May 1992	1.3 (0.5)[c]	29.1 (1.3)[c]	No Data	No Data
Separations				
11 June 1992	2.1 (1.9)	6.6 (1.3)	4.89	4.26
7 August	3.4 (1.2)	5.2 (0.7)	3.76	3.44
15 October	2.6 (1.5)	8.2 (1.6)	4.21	4.13
30 March 1993	1.6 (0.7)	No Data		
4 May	8.2 (1.7)	2.4 (2.0)	4.19	3.47
6 June	12.0	5.5	3.54	3.32
22 October	No Data	No Data	3.30	3.16

[a] For 27 May 1992 and 11 June 1992, d.f.=5. For all other dates, d.f.=2.
[b] From Swift *et al.* (1993).
[c] Clover as % ground cover.

The N content of clover residues incorporated in 1992 was significantly higher than that of the grass residues from both the grass ($p<0.001$) and grass-clover ($p<0.05$) swards. The same differences were observed on both swards in 1993 ($p<0.01$).

The N content of grass herbage showed a general downward trend over the duration of the trial (Figure 4.1.6.1). In the undisturbed treatments, the N content of grass was generally *ca.* 3% N in 1992, with the exception of the undisturbed grass sward on 7 August. By 6 June 1993 the grass N content in the undisturbed swards had fallen to *ca.* 1.85% N.

The N content of herbage from the resown 1992 treatments (16 September 1992) was significantly higher than that of the previous cut (7 August) from the undisturbed treatments ($p<0.05$). The N content of the resown 1992 herbage was significantly higher than that of the undisturbed swards on 15 October ($p<0.01$). The resown 1992 herbage N content subsequently decreased to 1.5-1.6% N by 6 June 1993, significantly lower than the undisturbed swards ($p<0.05$).

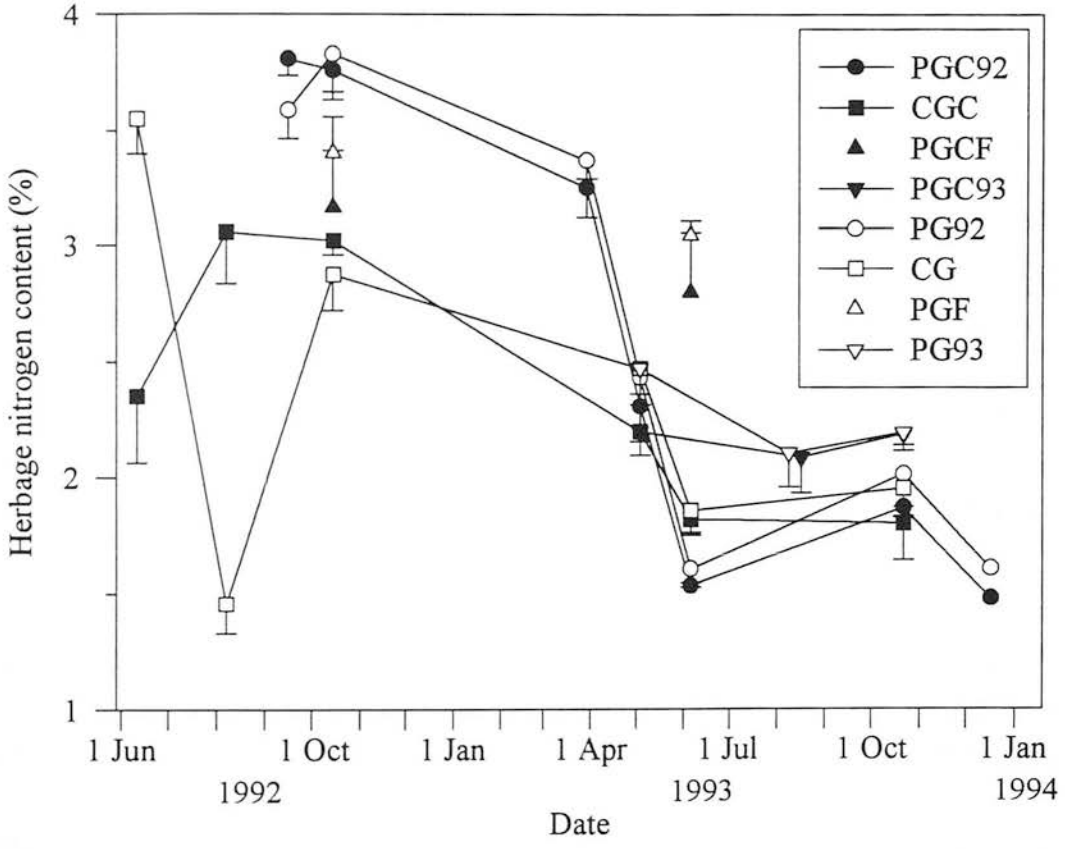

Figure 4.1.6.1 Nitrogen content of above ground herbage (treatment means) on the Beechgrove field trial, 1992-3 (Bars = SE, d.f.=2).

In 1993, the N content of herbage from the resown 1993 treatments did not fall as much as that from the undisturbed swards. Consequently, on 22 October 1993 the herbage N content in the resown 1993 treatments was significantly higher than that of the undisturbed treatments ($p<0.05$), though not quite significantly higher than that of the resown 1992 ($0.05<p<0.1$).

On 15 October 1992 the N content of regrowth from the fallow treatments was not quite significantly higher than the grass in the undisturbed swards ($0.05<p<0.1$), and on 6 June 1993 was higher than that of herbage from all other treatments ($p<0.05$).

Peak DM production rates from the undisturbed swards were somewhat higher in 1992 (7 August) than in 1993 (6 June). This contrast between the two seasons was more apparent in N uptake rates in herbage (Figure 4.1.6.2).

On 7 August 1992, there was no significant difference between DM production on the two sward types. However, the lower N content of herbage on the undisturbed grass sward led to N uptake which was significantly lower than that by the undisturbed grass-clover sward between 7 August and 15 October 1992 ($p<0.01$). In 1993, N uptake by the grass sward between 9 August and 22 October was not quite significantly greater than by the grass-clover sward ($0.05<p<0.1$).

Between 16 September and 15 October 1992, N uptake by herbage on the resown 1992 treatments was significantly greater than that from the undisturbed treatments ($p<0.001$), and remained so until 4 May 1993 ($p<0.05$). Between 6 June and 9 August 1993 N uptake by the undisturbed swards was more than that by the resown 1992 swards ($p<0.05$) as DM production rates on the latter treatment fell faster.

In 1993, between their first cut and 22 October, the resown 1993 swards showed greater herbage N uptake than the undisturbed swards ($p<0.001$).

In both years, there was never a significant effect of sward type on the resown treatments herbage N uptake. However, N uptake by regrowth on the grass-clover fallow treatment was greater than that on the grass fallow treatment ($p<0.01$).

When all above ground herbage was sampled on 17 December 1993, the DM from the resown 1993 treatments was significantly lower than from the undisturbed and resown 1992 treatments ($p<0.01$).

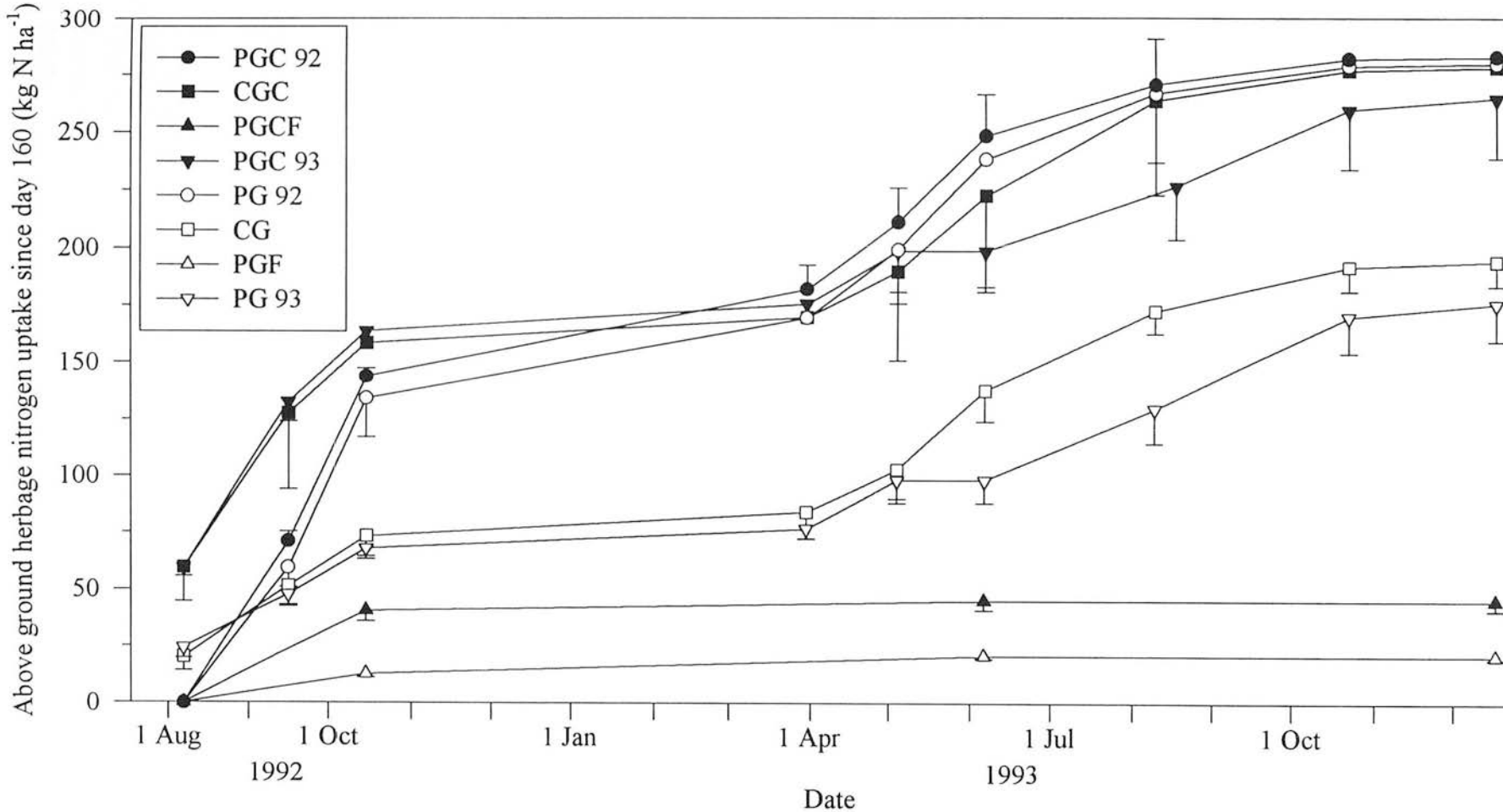

Figure 4.1.6.2 Cumulative nitrogen offtake by above ground herbage[a] (treatment means) on the Beechgrove field trial, 1992-3 (Bars = SE)[b].

[a] See Appendix 2 for details of assumptions made in calculating nitrogen uptake by herbage.

[b] D.f.=2. Where error bars are not shown, bars from the nearest sampling dates in the same treatment are representative.

4.1.7 Estimation of mineral nitrogen release on the Beechgrove trial

The estimation of mineral N by addition of estimates of several N processes presents many potential problems. The fallow treatment would appear to be the easiest treatment in which to estimate total N release due to the limited plant uptake (Table 4.1.7). However, gaseous N loss may be vastly underestimated in this treatment due to the limited sampling period, particularly in 1992. Prior to the onset of drainage, the soil was left practically bare and would still have contained considerable amounts of mineral N. Perhaps more importantly, there is considerable doubt about the leaching estimate from this treatment (section 5.5).

Table 4.1.7 Estimation of mineral nitrogen release (kg N ha^{-1}) between 18 June 1992 and 17 December 1993 on the Beechgrove trial using measured soil mineral nitrogen, plant uptake, gaseous nitrogen and leaching losses.

Nitrogen pool	Estimated nitrogen flux / content (kg N ha^{-1})					
Treatment	PGC 92	PG 92	CGC	CG	PGCF	PGF
Soil mineral nitrogen content (0-20 cm) on 18 June 1992	28.6	21.2	27.1	25.2	28.6	21.2
Final soil mineral nitrogen content (0-20 cm)[a]	6.8	7.4	30.2	47.2	22.0	18.1
Above ground plant uptake[b]	283.5	280.5	278.9	194.3	44.8[c]	20.7[c]
Root uptake[d]	44.4	39.2	0	0	10.4[c]	4.4[c]
Denitrification[e]	17.8	8.2	2.1	1.2	21.6	10.5
Leaching[f]	49.9	44.3	3.9	0.7	229.3	441.2
Total mineral N released	373.8	358.4	288.0	218.2	244.3	448.6

[a] 21 September and 11 October 1993 for the resown 1992 and undisturbed treatments, respectively. For the fallow treatments, an estimate for 17 December is derived by linear interpolation between 11 October 1993 and 20 January 1994.
[b] For the resown 1992 treatments this includes all above ground herbage, not just that above cutting height. For the undisturbed swards, herbage above ground, but below the cutting height, was assumed to be stable.
[c] Since all regrowth on the fallow treatments was killed and left on or within the soil, these figures are not included in the total estimate of nitrogen release.
[d] See Appendix 2.7 for details of estimation.
[e] See sections 3.8.1 and 4.1.4 for details.
[f] For the resown 1992 and undisturbed treatments, leaching from the onset of drainage in 1993 to 17 December 1993 is excluded since load estimates were not greater than 1 kg N ha^{-1}.

The resown 1992 treatments were considered to provide a more reliable estimate of mineral nitrogen release than the fallow treatments for several reasons. Firstly, the resown sward would 'mop up' soil mineral nitrogen quickly, reducing the potential for gaseous N loss beyond the sampling period. Secondly, plant N uptake was greater in this treatment and, given the greater sampling area, was probably more reliable than leaching or gaseous N loss estimates. Finally, due to the anticipated lower leaching load, quantitative inaccuracies in leaching load estimates would be smaller.

4.2 GLENCORSE FIELD TRIAL

4.2.1 Sward residue inputs

Table 4.2.1 Dry matter quantity and N content of sward residues (treatment means) for the Glencorse trial, 1993 (Standard errors in parentheses, d.f.=1).

Residue	Treatment	DM (tonnes ha^{-1})	Residue N content (% N)	Residue N input (kg N ha^{-1})
Plant tops + stubble	PC	8.03 (1.31)	0.99 (0.18)	12.6 (37.5)
	PG	6.09 (0.81)	0.80 (0.09)	52.4 (11.6)
MOM, 0-4 cm	PC	4.65 (0.73)	0.92 (0.12)	43.2 (12.0)
	PG	6.62 (0.19)	0.70 (0.02)	45.8 (0.5)
MOM, 4-20 cm	PC	4.15 (1.45)	0.98 (0.05)	41.1 (16.0)
	PG	5.38 (1.52)	0.90 (0.06)	49.2 (16.7)
MOM, 20-40 cm	PC	1.08 (0.18)	0.98[a]	10.4 (1.2)
	PG	1.28 (0.02)	0.90[a]	11.5 (1.0)
TOTAL	PC	17.92 (3.31)		207.3 (64.3)
	PG	19.37 (0.93)		158.9 (6.6)

[a] Nitrogen content assumed to be the same as 4-20 cm sample.

The clover content of the grass-clover treatment (PC), 17.8% of total DM (SE 4.9), was almost significantly higher than that of the grass treatment (PG), 2.4% of total DM (SE 0.1) ($0.05<p<0.1$). This treatment effect was mainly due to the very high clover content of plot 8, 22.6% (SE 3.1), which had a significantly higher clover content than both of the grass plots ($p<0.05$).

The N content of plant tops in plot 4 was significantly higher than in plot 5 ($p<0.01$). The N content of clover, 3.12% N (SE 0.07), was greater than that of grass in both treatments ($p<0.01$).

Plot 8 had a significantly higher tops DM input than plots 1 and 5 ($p<0.05$), but there was no significant treatment effect. There were no significant differences in the total DM inputs.

The DM input from macro-organic matter was always greater than that from plant tops, particularly in the grass treatment, but there was never a significant difference in N input from these fractions.

There was no significant difference in total root N inputs, but inputs from plots 5 and 8 were much higher than from plots 1 and 4. Plot 8 had significantly higher N input from plant tops than plots 4 and 5 ($p<0.05$) and, significantly higher total N input than plot 4 ($p<0.05$). N input from plant tops and total N input on plot 1 were not quite significantly lower than on plot 8 ($0.05<p<0.1$).

4.2.2 Soil mineral nitrogen

On 29 June 1993, prior to ploughing, the high residue treatments (PC and PG) had significantly higher NH_4^+-N ($p<0.05$), but significantly lower NO_3^--N concentrations ($p<0.01$) than the previously arable, low residue treatments (AN and CN). There was no significant difference in mineral N concentrations between the treatments (Figure 4.2.2.4).

Following the ploughing of the swards on 14 July 1993, NH_4^+-N concentrations in all plots, except plot 8, showed a sharp decline. On 19 July, NH_4^+-N concentrations in plot 8 were significantly higher than in all other plots ($p<0.01$). Nitrate-N concentrations in the grass (PG) and grass-clover (PC) treatments increased but remained lower than in the low residue treatments.

Nitrate-N concentrations rose steadily in all treatments prior to fertilisation, especially in plot 8, the only plot to show a marked increase in mineral N between ploughing and 4 August. By 28 July mineral N concentrations in plot 8 were significantly higher than in all other plots ($p<0.01$), and concentrations in plots 1 and 5 were significantly lower than in all other plots ($p<0.05$).

Between ploughing and fertilisation, the high residue treatments continued to have higher NH_4^+-N concentrations than the low residue treatments, significant on 12 August ($p<0.05$). Between ploughing and fertilising NO_3^--N concentrations in the PG treatment were not quite significantly lower than the CN treatment ($0.05<p<0.1$). Immediately prior to fertilising, mineral N concentrations in the PC treatment were higher than in the other treatments and concentrations in plot 8 were significantly higher than in all plots except 1 and 3 ($p<0.05$).

Following fertiliser application, NH_4^+-N concentrations in the AN treatment were significantly higher than in the CN and PC treatments on 19 August ($p<0.001$). Treatments fertilised with calcium nitrate showed significantly higher NO_3^--N concentrations than the other treatments on 26 August ($p<0.05$).

On 19 August, mineral N concentrations in the fertilised treatments were significantly higher than in the unfertilised PC treatment ($p<0.05$), and concentrations in plot 1 were significantly lower than in all plots except 2 and 5 ($p<0.05$) due primarily to its lower NO_3^--N concentrations. In contrast, NO_3^--N concentrations in plot 8 were not significantly lower than any of the fertilised plots on 19 August. On 26 August mineral N concentrations peaked in the PG treatment, one week later than in all other treatments, and were significantly higher than in treatment CN ($p<0.05$), though not quite significantly higher than in treatment PC ($0.05<p<0.1$).

Due to the large variability after fertilising, NH_4^+-N concentrations in the AN treatment were only significantly higher than in the CN treatment on 8 October ($p<0.01$). In the PG treatment NH_4^+-N concentrations increased steadily between 19 July and 2 September. Plot mean data showed that NH_4^+-N concentrations in plots 4 and 5 (PG) were significantly higher than in plots 6 and 7 (CN) on 26 August ($p<0.05$). Ammonium-N concentrations in the PC treatment were also significantly higher than in the CN treatment on 19 ($p<0.001$) and 26 August ($p<0.05$).

After 26 August, all treatments showed a general pattern of falling NO_3^--N concentrations until 8 October. During this period the PC treatment tended to have the lowest mineral N concentration, primarily due the low concentrations in plot 1. Nitrate-N concentrations in the PC treatment were significantly lower than in the CN treatment on 20 September and 8 October, and the AN treatment on 20 September ($p<0.05$). After 26 August, there was never a significant difference between mineral N concentrations in the AN and CN treatments.

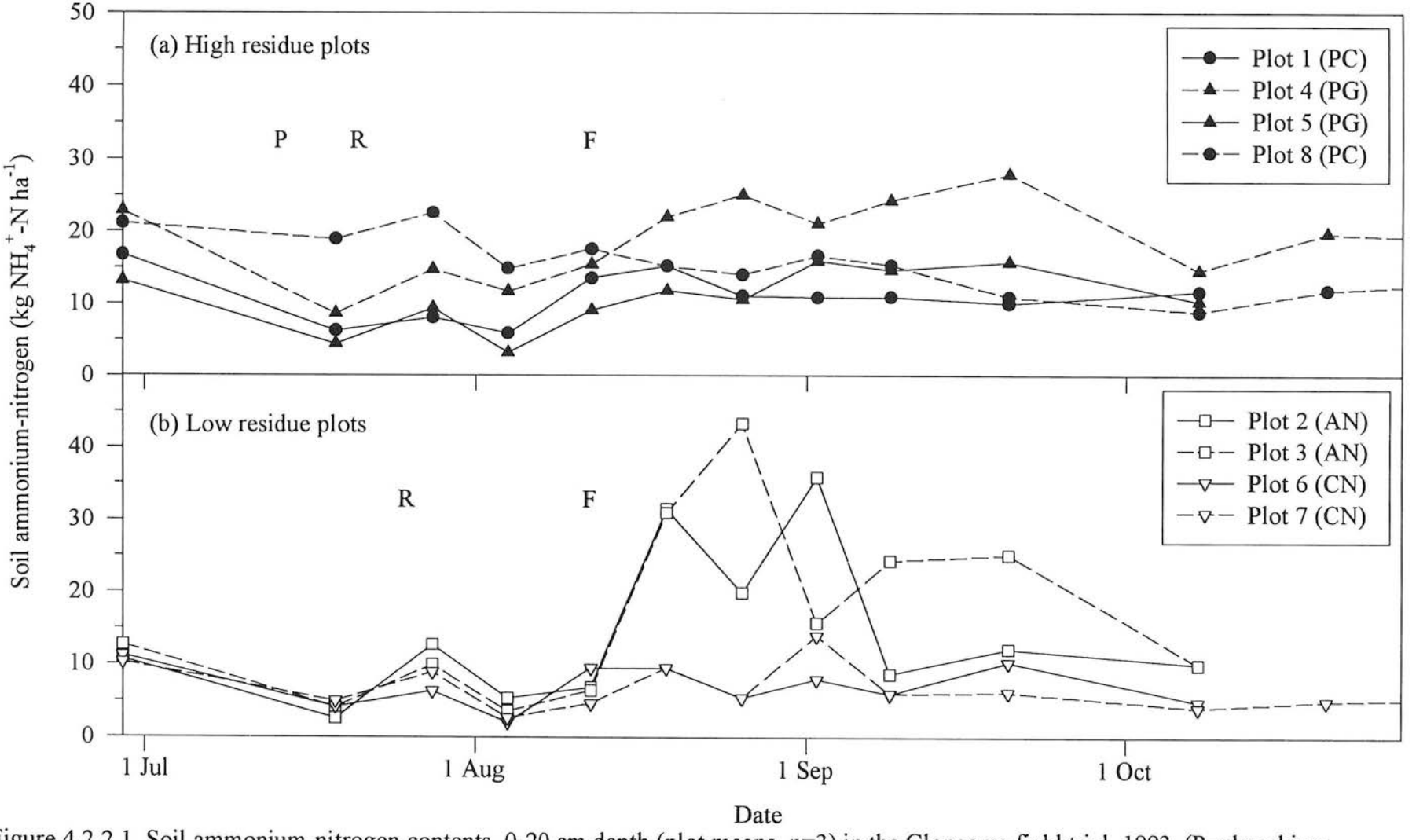

Figure 4.2.2.1 Soil ammonium-nitrogen contents, 0-20 cm depth (plot means, n=3) in the Glencorse field trial, 1993. (P=ploughing, R=rotavation, F=fertiliser application)

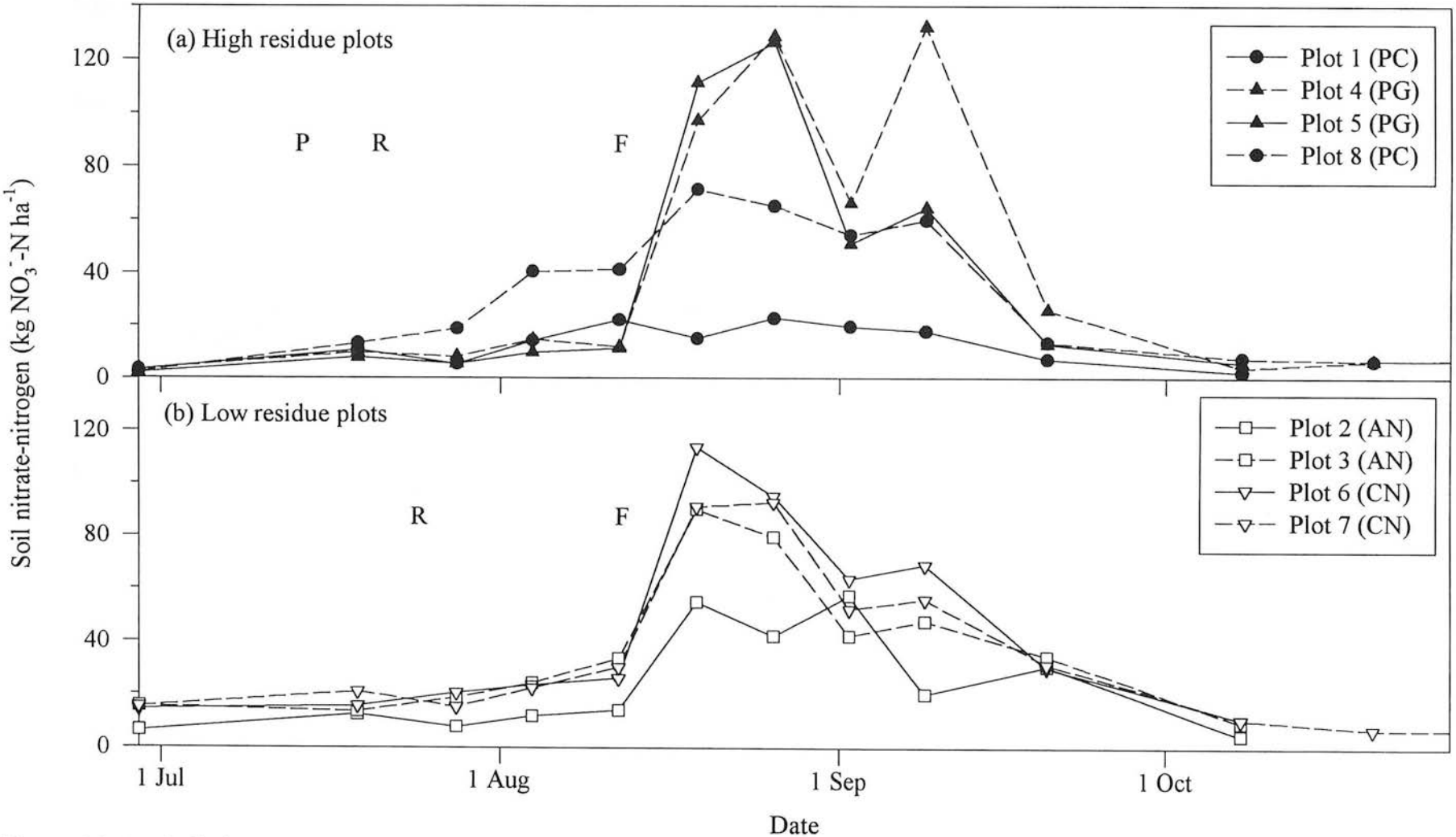

Figure 4.2.2.2 Soil nitrate-nitrogen contents, 0-20 cm depth (plot means, n=3) in the Glencorse field trial, 1993. (P=ploughing, R=rotavation, F=fertiliser application)

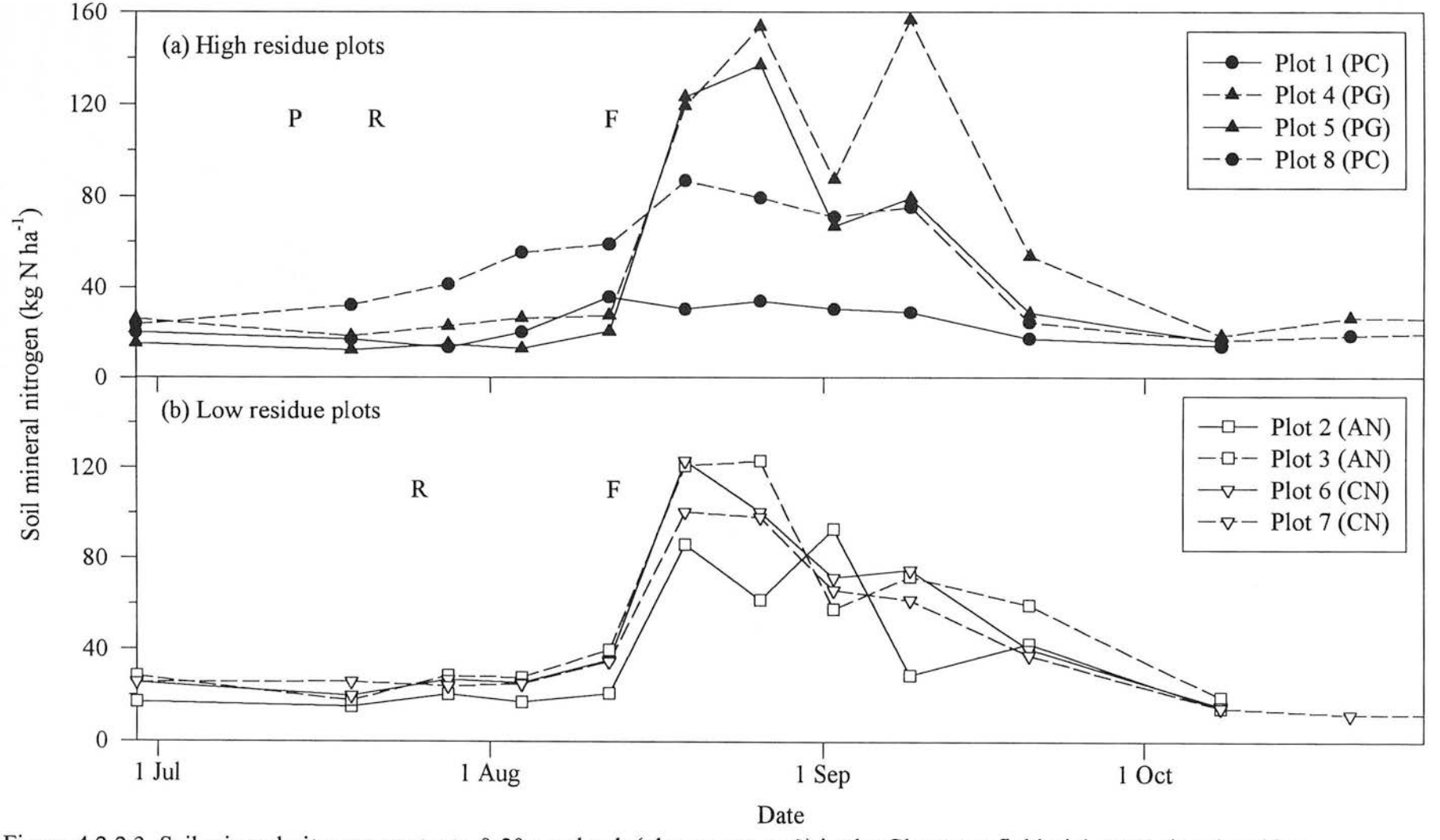

Figure 4.2.2.3 Soil mineral nitrogen contents, 0-20 cm depth (plot means, n=3) in the Glencorse field trial, 1993. (P=ploughing, R=rotavation, F=fertiliser application)

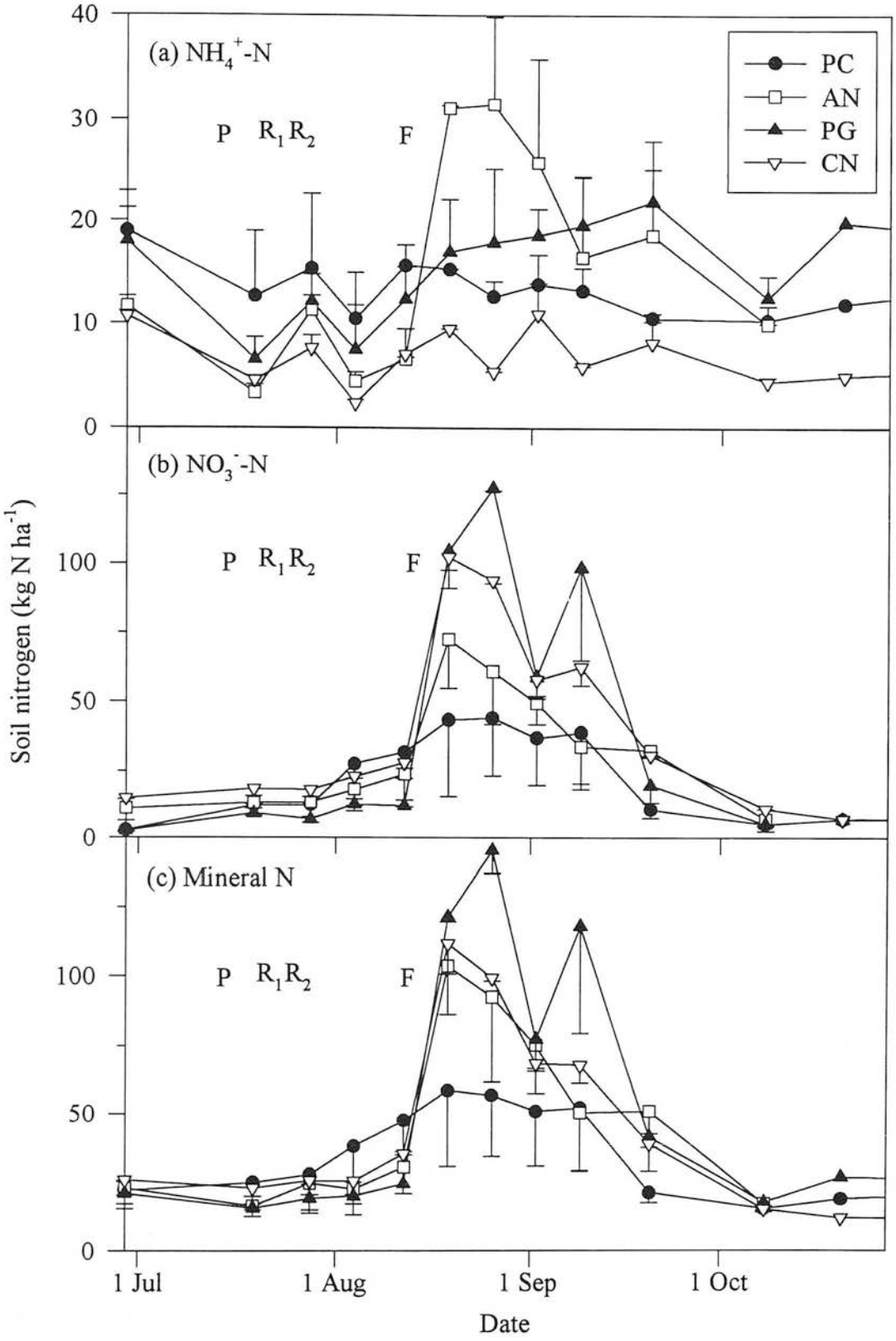

Figure 4.2.2.4 Soil ammonium-nitrogen, nitrate-nitrogen and mineral nitrogen contents, 0-20 cm depth (treatment means) in the Glencorse field trial, 1993 (Bars = SE, d.f.=1). (P=ploughing, R_1=rotavation of PG and PC treatments, R_2=rotavation of AN and CN treatments, F=fertiliser application)

4.2.3 Nitrous oxide fluxes

The patterns of N_2O fluxes are shown in Figure 4.2.3. For treatments AN and CN only one plot was sampled, plots 2 and 7, respectively. Hence it was not possible to make statistical comparisons between treatments AN and CN and the other treatments. Differences between PG and PC treatments were tested using analysis of variance whilst t-tests were used for comparison with plots 2 and 7. Differences between plots 2 (AN) and 7 (CN) were analysed using duplicate boxes from within each plot.

Throughout the field trial there was no significant difference between those treatments receiving high residue inputs (PG and PC) and low residue inputs (CN and AN), as assessed by two-way analysis of variance. However, on most occasions the high residue treatments produced higher fluxes.

On no occasion were the N_2O fluxes from the fertilised treatments (PG, CN and AN) significantly greater than from the unfertilised PC treatment.

Six days after ploughing (20 July) all treatments gave low fluxes. The flux from the grass-clover treatment was not quite significantly greater than from the other treatments ($0.05<p<0.1$).

Following rotavation (21 July treatments PC and PG, 26 July treatments AN and CN) only the grass-clover treatment showed an immediate increase in fluxes. Emissions from plot 8 were significantly higher than all other plots ($p<0.01$). On 2 August fluxes from the PC treatment were greater than from the PG treatment and plots 2 (AN) and 7 (CN). Fluxes from plot 7 (CN) were frequently significantly lower than from the other plots ($p<0.05$).

Following fertiliser application on 12 August fluxes from plot 4 increased sharply. Plots 7 (CN) and 2 (AN) showed no immediate reaction to fertiliser application and emissions were significantly lower than from all other plots on 17 August ($p<0.05$ and 0.01, respectively). This pattern remained on 19 August, as high fluxes were sustained from high residue treatments.

After 2 September, 50 days after ploughing, the main period of flux activity occurred. Emissions increased rapidly until 17 September from the grass-clover treatment, and

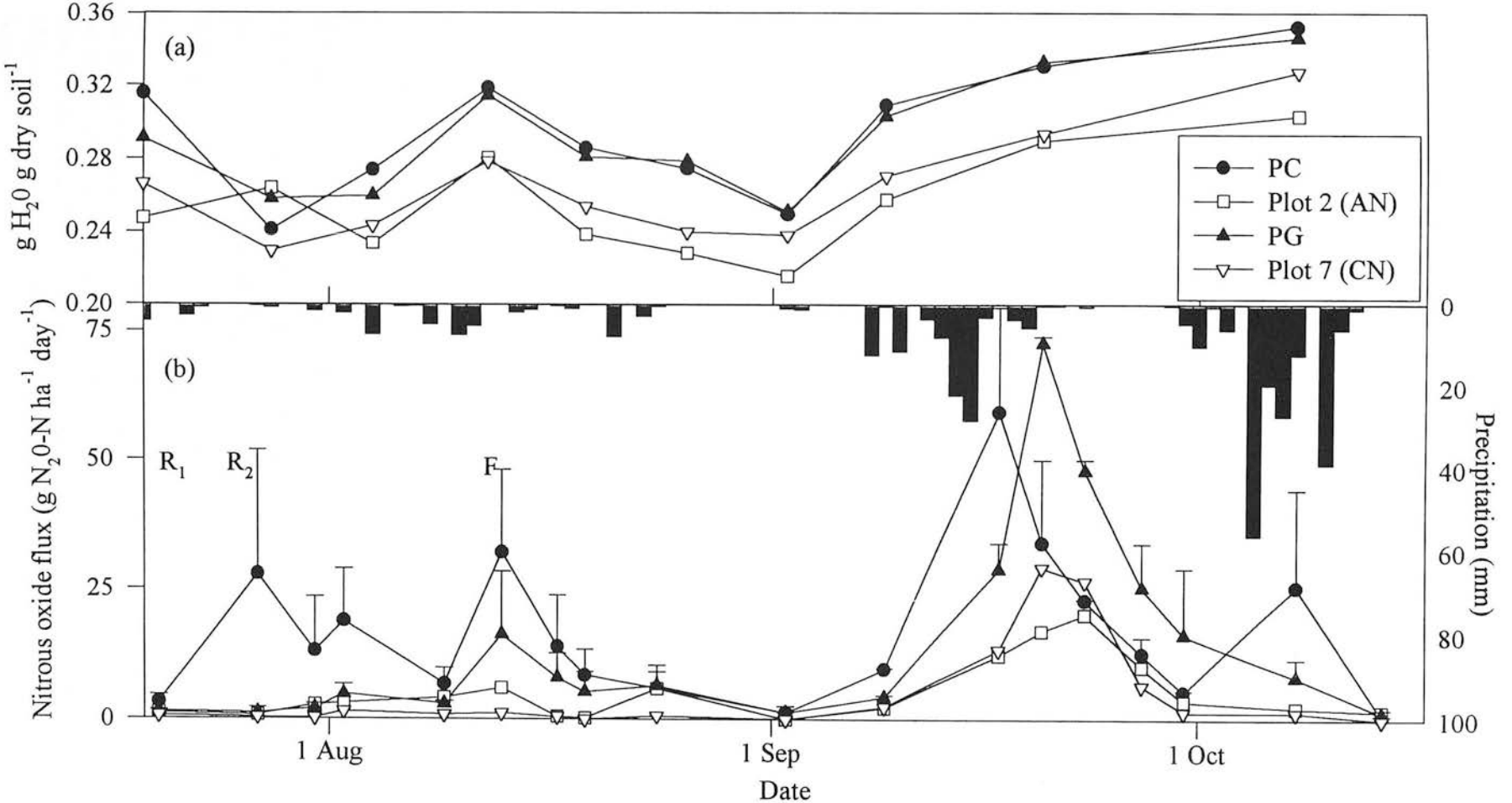

Figure 4.2.3 (a) Soil gravimetric moisture contents (treatment means). (b) Nitrous oxide fluxes (treatment means[a]) and daily precipitation in the Glencorse field trial, 1993 (Bars = SE, d.f.=1).

(R_1 and R_2=rotavation of treatments PG and PC, and AN and CN, respectively. F=fertiliser application)

[a] Measurements were only taken from one plot in the AN and CN treatments. SE not given.

until 20 September for all other treatments. The grass-clover treatment showed significantly higher fluxes than the fertilised plots on 9 September ($p<0.05$). The grass treatment had significantly higher fluxes than plots 2 (AN) and 7 (CN) on 20 ($p<0.01$) and 23 September ($p<0.05$). Only on 23 September was the grass treatment significantly greater than the grass-clover treatment ($p<0.01$).

Fluxes declined to very low values by 14 October, with the exception of increased fluxes on 8 October from the grass-clover treatment and plot 5 (PG).

Table 4.2.3 Cumulative N_2O fluxes (plot and treatment means[a]) from the Glencorse trial between 20 July and 11 November,1993 (Standard errors in parentheses; d.f.=1).

Treatment (Plot)	Nitrous Oxide Flux (kg N_2O-N ha^{-1})	Denitrification flux[b] (kg N_2-N + N_2O-N ha^{-1})
PC	1.5 (0.3)	8.0 (1.9)
PG	1.0 (0.2)	5.7 (1.1)
AN (2)	0.4 (0.0)	2.3 (0.2)
CN (7)	0.3 (0.1)	1.8 (0.4)
1	1.1 (0.3)	6.1 (1.8)
8	1.8 (0.0)	9.9 (0.2)
4	1.3 (0.4)	6.8 (2.4)
5	0.8 (0.2)	4.6 (1.1)

[a] Treatment means = 2 plots per treatment, plot means = 2 replicates per plot.
[b] Calculated using the average N_2+N_2O:N_2O ratio for the ungrazed sward in the incubation experiment, 5.4 (see section 5.4.2).

4.2.4 Leaching losses

4.2.4.1 Soil water nitrogen concentrations in porous ceramic cups

The first soil solution samples were taken on 21 September 1993 (Figure 4.2.4.1)[3]. On 24 September plot 8 showed significantly higher NO_3^--N concentrations than plot 7 ($p<0.05$). However, in plot 7 the fall in NO_3^--N concentrations was slower and by 15 October concentrations were higher than those in plot 4, but remained

[3]In Figure 4.2.4.1 and 4.2.4.2 soil water nitrogen concentration data for porous cups are plotted on the closest well/drain sampling date and the text may therefore appear to disagree with the figure. This only occurs on three occasions: 17 September 1993, 27 September 1993 and 4 March 1994 when cups were actually sampled on 21 September 1993, 24 September 1993 and 2 March 1994, respectively.

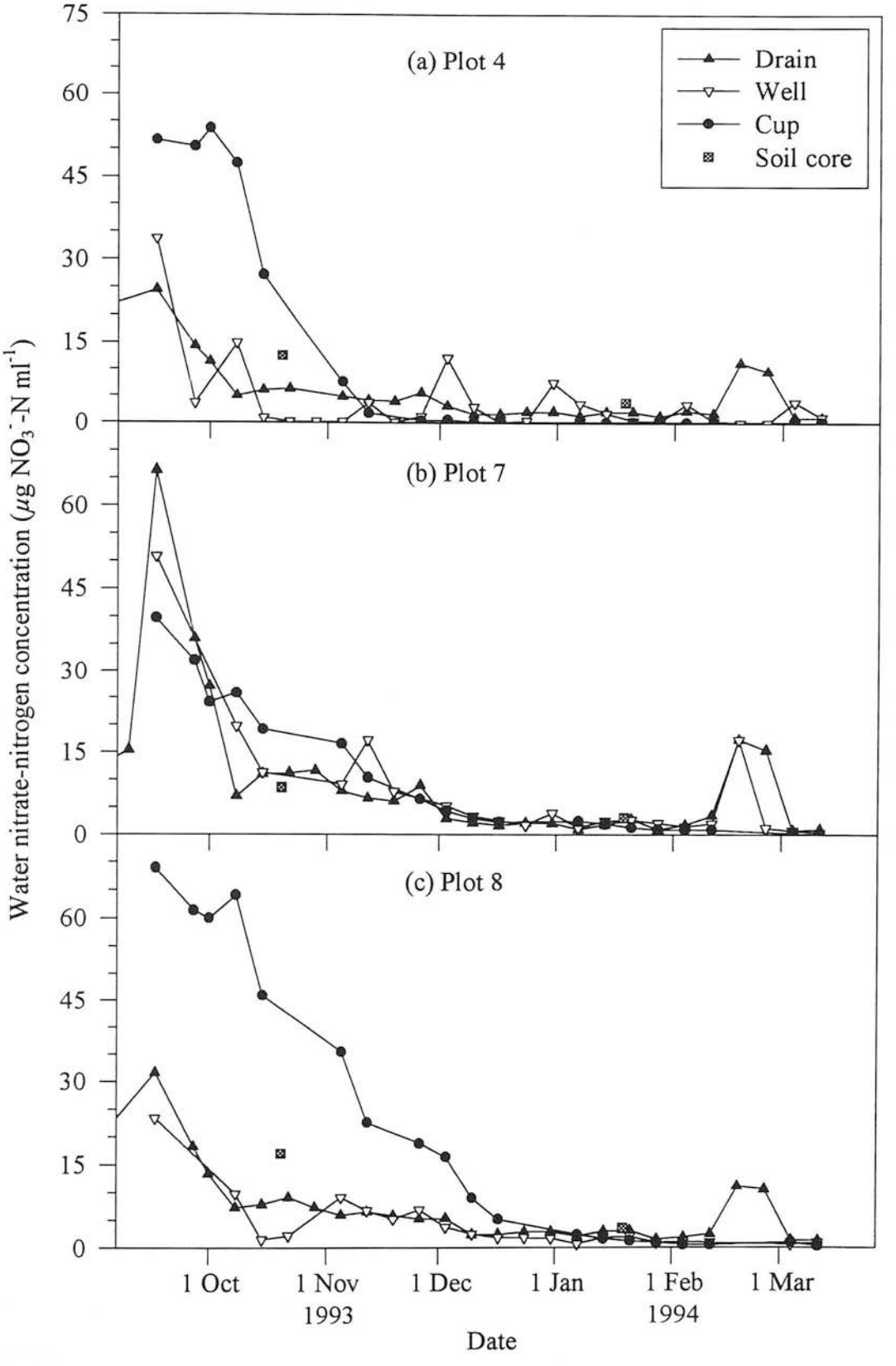

Figure 4.2.4.1 Nitrate-nitrogen concentrations in samples from porous cups[a], wells[a], drains[b] and soil cores[a] at the Glencorse field trial, 1993-4.
[a] n=2, [b] n=1.

significantly lower than those in plot 8 ($p<0.05$). Nitrate-N concentrations in plot 4 were significantly lower than in plot 8 on 15 October ($p<0.05$), 12 ($p<0.01$) and 26 November ($p<0.001$), and remained below 1 μg NO_3^--N ml^{-1} thereafter. Whilst NO_3^--N concentrations continued to fall in plots 7 and 8, the decline was more exponential, with plot 8 remaining higher until 7 January 1994. Thereafter NO_3^--N concentrations in plots 7 and 8 remained very similar until sampling ceased.

4.2.4.2 Calculated leaching losses

Table 4.2.4.2 Calculated nitrate-N leaching loads from the Glencorse field trial over winter 1993-4[a] using plot drainage volumes in combination with (a) drainage water concentrations and (b) porous cup sample concentrations (Standard errors in parentheses, d.f.=1).

	Nitrate-N leaching load (kg N ha^{-1})	
Treatment / Plot	(a)	(b)
1	13.0	-
2	20.1	-
3	34.6	-
4	28.3	70.0 (16.6)
5	23.7	-
6	50.1	-
7	58.2	57.0 (17.4)
8	38.1	118.1 (10.9)
PC	25.5 (12.5)	-
AN	27.4 (7.3)	-
PG	26.0 (2.3)	-
CN	54.1 (4.1)	-

[a] Calculated for the period 14 September 1993 to 11 March 1994.

4.2.4.3 Comparison of soil water sampling methods

4.2.4.3.1 Porous cup residual samples vs. porous cup suction samples

There were no significant differences in the NO_3^--N concentrations of the residual water collected from the porous cups and that collected using a -20 kPa falling suction. In plot 8 the residual water tended to be slightly higher on each sampling date. This pattern may have been an artefact of the falling NO_3^--N concentrations since the residual sample was taken prior to the suction sample on each occasion.

This comparison only took place between 17 December 1993 and 21 January 1994, when NO_3^--N concentrations were very low, and a clearer picture may have been gained earlier in the drainage season.

4.2.4.3.2 Porous cup suction samples vs. drains

Whilst data was very limited, t-tests were possible to compare the concentrations of NO_3^--N in drain water with that of water sampled using porous ceramic cups (Figure 4.2.4.1).

Nitrate-N concentrations in porous cup samples from 24 September were higher than drain water samples taken on 27 September, significantly so in plots 4 ($p<0.05$) and 8 ($p<0.01$). The subsequent decline in NO_3^--N concentrations was less abrupt in porous cup samples than in drain water. Consequently, in plots 4 and 8 the NO_3^--N concentration in drain water remained significantly lower than that in cup samples until 5 November ($p<0.05$), with the exception of 1 October in plot 4 ($0.05<p<0.1$). Similarly, in plot 7, by 8 October porous cup water had significantly higher NO_3^--N concentrations than drain water ($p<0.05$).

Thereafter, NO_3^--N concentrations in porous cup samples fell faster than drain water samples, the latter rarely falling below 1 μg NO_3^--N ml^{-1}. On 12 November in plot 4 the NO_3^--N concentration in drain water was significantly higher than that in porous cup samples ($p<0.05$), the difference being more significant thereafter ($p<0.01$).

After 28 January 1994, NO_3^--N concentrations in drain water tended to increase, particularly on 18 and 25 February. Nitrate-N concentrations in porous cup samples remained below 1 μg NO_3^--N ml^{-1}, significantly lower than drain water for all plots on 11 March 1994 ($p<0.05$). Unfortunately, on 18 and 25 February, porous cups failed to yield samples and thus no comparison was possible.

4.2.4.3.3 Porous cup suction samples vs. wells

Comparison of soil water NO_3^--N concentrations from wells and cups was carried out using two-way analysis of variance where possible, and one-way analysis of variance elsewhere. Plot means for each sampling date are shown in Figure 4.2.4.1.

In plots 4 and 8 the temporal pattern of NO_3^--N concentrations shown by the well samples was more erratic than that of the porous cup samples.

A two-way analysis of variance comparing well samples from 17 September with porous cup samples from 21 September revealed that NO_3^--N concentrations in cup samples were significantly higher than in well samples ($p<0.05$). This difference between the two sampling methods remained, where two-way analysis of variance was possible, until 12 November ($p<0.05$). On no occasion did one way analysis of variance of well and cup samples NO_3^--N concentrations show a significant difference in plot 7.

Between 12 November 1993 and 2 March 1994 there were no significant differences between sampling methods. On 11 March 1994, NO_3^--N concentrations in well samples were not quite significantly higher than in the porous cup samples ($0.05<p<0.1$).

4.2.4.3.4 Porous cup suction samples vs. wells (NH_4^+-N concentration)

Unlike all other sampling techniques, NH_4^+-N concentrations detected in well samples were frequently above the reliable detection limit (Figure 4.2.4.3). Two-way analysis of variance showed that the well samples had significantly higher NH_4^+-N concentrations than the porous cup samples on 5 November, 7 ($p<0.01$) and 28 January 1994 ($p<0.05$).

4.2.5 Plant dry matter production and nitrogen uptake

The AN treatment had the highest N uptake, significantly more than the CN treatment ($p<0.05$) (Table 4.2.5). Plots 1 ($p<0.05$) and 5 ($0.05<p<0.1$) had lower total N uptake than plots 2, 3, 4 and 8 due to poorer grass establishment.

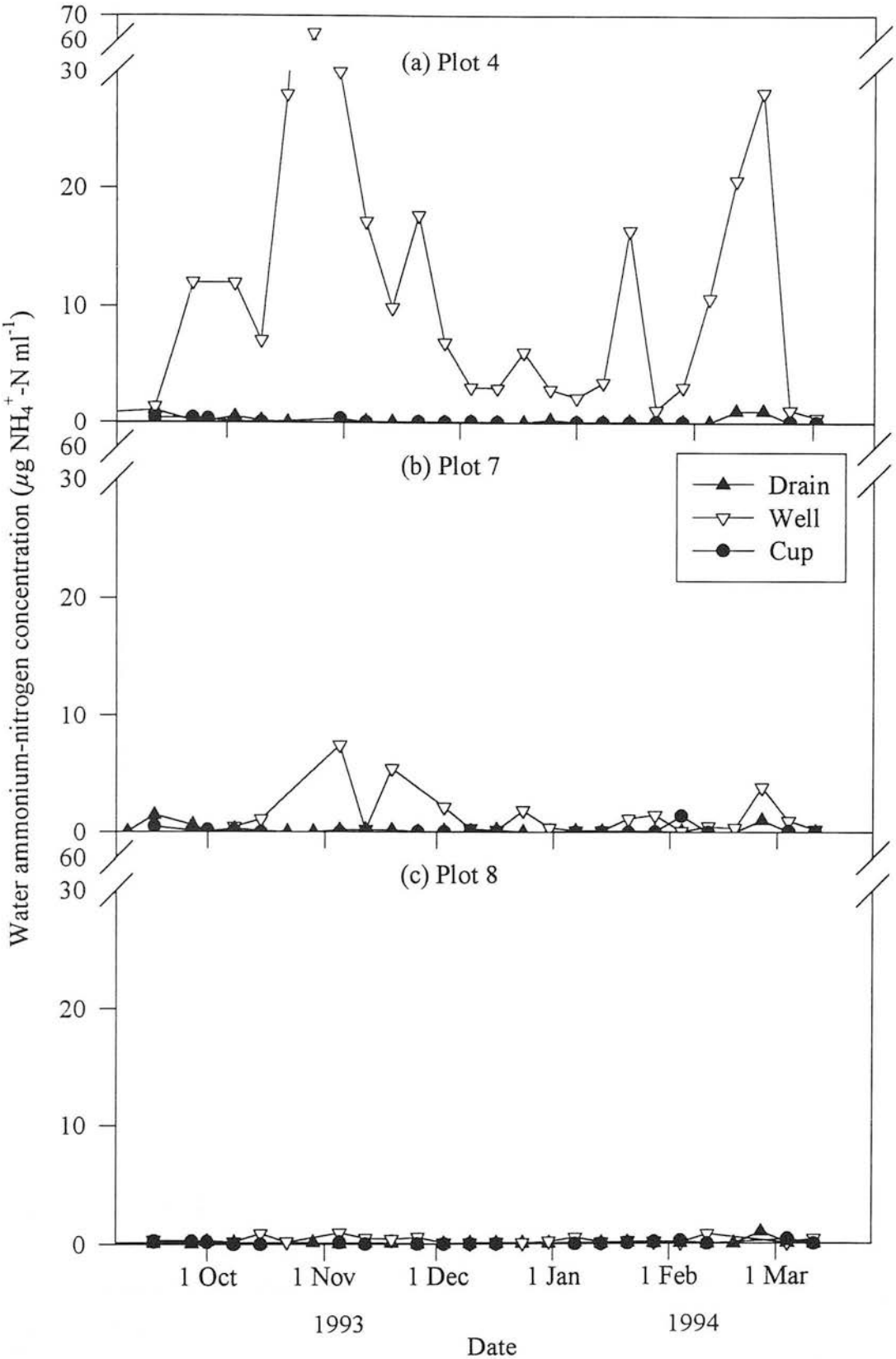

Figure 4.2.4.3 Ammonium-nitrogen concentrations in water samples from porous cups[a], wells[a] and drains[b] at the Glencorse field trial, 1993-4.
[a] n=2, [b] n=1.

Table 4.2.5 Uptake of nitrogen by resown grass (including roots[a]) on the Glencorse field trial, 1993-4 (Standard errors in parentheses, d.f.=1).

Treatment / Plot	Grass tops N content (% N)	Nitrogen Uptake (kg N ha^{-1})
1	2.74	23.10 (3.87)
2	2.38	51.08 (6.81)
3	2.54	55.7 (11.9)
4	2.93	57.3 (11.9)
5	2.35	27.97 (3.22)
6	2.71	38.33 (9.84)
7	2.51	35.98 (6.62)
8	3.18	60.2 (13.0)
PC	2.96 (0.22)	41.7 (18.6)
AN	2.46 (0.08)	53.39 (2.31)
PG	2.64 (0.29)	42.6 (14.7)
CN	2.61 (0.10)	37.16 (1.18)

[a] This assumes root DM is equivalent to half that of the tops and uses a measured N content for roots of 1.37% N.

4.3 COWLOAN FIELD TRIAL

4.3.1 Nitrate-nitrogen concentrations in porous cup samples

Nitrate-N concentrations in the ploughed out grass-clover treatment (OGC) reached a peak of 7.4 μg NO_3^--N ml^{-1} on 6 November 1992 (Figure 4.3.1). After this date NO_3^--N concentrations in all treatments showed a decreasing trend until 4 December. Despite NO_3^--N concentrations being considerably higher in the OGC treatment than in all the other treatments, due to variability it was never significantly higher than the undisturbed grass-clover treatment (CGC). On 29 October and 6 November NO_3^--N concentrations in the ploughed out grass treatment (OPG) were significantly higher than in the CGC treatment ($p<0.05$).

The previously arable treatment (ARA) showed NO_3^--N concentrations significantly higher than in the CGC treatment on 15 ($p<0.05$), 22 ($p<0.05$) and 29 January 1993 ($p\leq0.001$), though not quite significantly higher than in the OPG treatment on 15 January 1993 ($0.05<p<0.1$). Nitrate-N concentrations in the OGC treatment were significantly higher than in the CGC treatment on 29 January 1994 ($p<0.05$), though not quite significantly higher than in the OPG treatment on 29 January and 5 February 1994 ($0.05<p<0.1$).

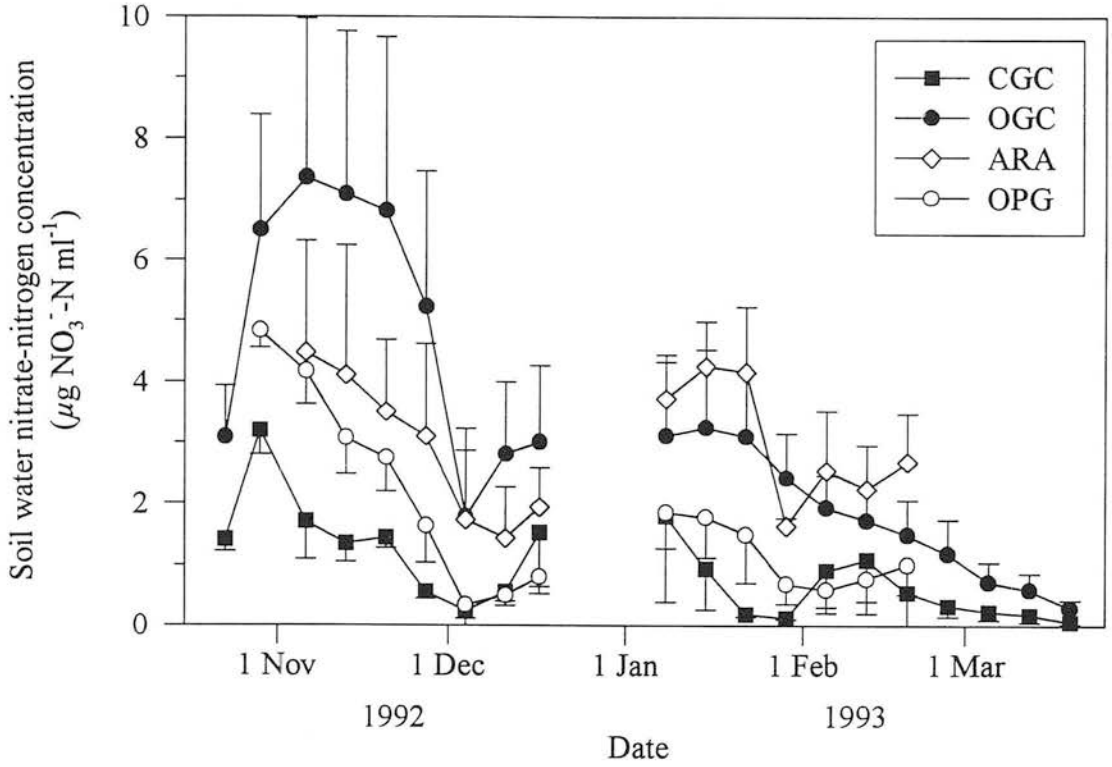

Figure 4.3.1 Nitrate-nitrogen concentrations (treatment means) in porous cup samples on the Cowloan trial, 1992-3 (Bars = SE, d.f.=2).

4.3.2 Leaching losses at the Cowloan field trial

Table 4.3.2 Estimated leaching losses (treatment means) between 12 October 1992 and 22 February 1993 on the Cowloan field trial (Standard errors of the mean given in parentheses, d.f.=2).

Treatment	Leaching loss (kg NO_3^--N ha^{-1})
Ploughed out 5 year old grass-clover ley, sown to spring barley (OGC)	12.8 (4.45)
Ploughed out 5 year old ryegrass ley, sown to spring barley (OPG)	6.1 (1.40)
Continued arable, sown to spring barley (ARA)	10.1 (3.12)
Undisturbed grass-clover ley (CGC)	3.1 (0.40)

4.4 LABORATORY EXPERIMENTS

4.4.1 Mineral nitrogen, carbon dioxide and gaseous nitrogen release following incorporation of grazed and ungrazed grass-clover swards

4.4.1.1 Sward residue inputs

Table 4.4.1.1 Mean dry matter quantity and N content of sward residues (Standard errors in parentheses, d.f.=2).

Residue	Treatment	DM ($g\ kg^{-1}$ dry soil)	N content	
			% N	kg N ha^{-1}
Plant tops	Grazed	0.83 (0.07)	1.44	27.72 (2.27)
	Ungrazed	0.94 (0.20)	1.31	28.80 (5.94)
MOM, 0-5 cm	Grazed	20.94 (1.73)	1.16	107.04 (3.42)
	Ungrazed	18.90 (0.70)	1.21	96.4 (2.65)
MOM, 5-20 cm	Grazed	3.66	1.06	67.67
	Ungrazed	3.96	1.11	76.89

There were no significant differences between residue inputs from the swards (Table 4.4.1.1).

4.4.1.2 Soluble organic carbon (SOC)

SOC concentrations in the grazed soil were not quite significantly greater than in the ungrazed soil ($0.05<p<0.1$) (Table 4.4.1.2). Using each turf replicate, SOC in the soil at day 0 showed no significant relationship with CO_2 evolution rates. SOC concentrations on day 53 were significantly less than on day 0 in both grazed ($p<0.01$) and ungrazed ($p<0.05$) treatments.

Table 4.4.1.2 Soluble organic carbon concentrations in the soil at the beginning and end of the incubation period (Standard errors in parentheses, d.f.=n-1).

	SOC (μg g dry soil^{-1})	
Treatment	Grazed	Ungrazed
Day 0 (n = 3)	138.1 (12.6)	94.6 (12.4)
Day 53 (n = 9)	43.08 (3.67)	49.05 (2.89)

4.4.1.3 Soil mineral nitrogen

During the first eight days after incorporation NH_4^+-N concentrations increased in both treatments and, in the grazed treatment, continued to increase until day 21. After day 28 NH_4^+-N concentrations became increasingly stable at approximately 22 kg NH_4^+-N ha^{-1} in both treatments (Figure 4.4.1.3a).

Nitrate-N concentrations increased steadily in both treatments during the first four weeks of incubation, and then increased more rapidly until day 53 (Figure 4.4.1.3b). Throughout the experiment NO_3^--N accumulated faster in the grazed treatment and, by day 21 NO_3^--N concentrations were significantly higher than in the ungrazed treatment ($p<0.05$).

Following the initial increase, mineral N concentrations in the ungrazed treatment fell slightly and remained stable at approximately 30 kg N ha^{-1} until day 39, after which a slight increase occurred (Figure 4.4.1.3c). In contrast, mineral N concentrations in the grazed treatment increased in roughly linear fashion throughout the incubation period and were significantly greater than the ungrazed treatment from day 39 onwards ($p<0.05$).

Two way analysis of variance on day 53 revealed significantly higher NH_4^+-N concentrations ($p<0.001$) and significantly lower NO_3^--N concentrations ($p<0.05$) in the ACET replicates than in the SOIL and GAS replicates, but no significant difference in total mineral N concentrations.

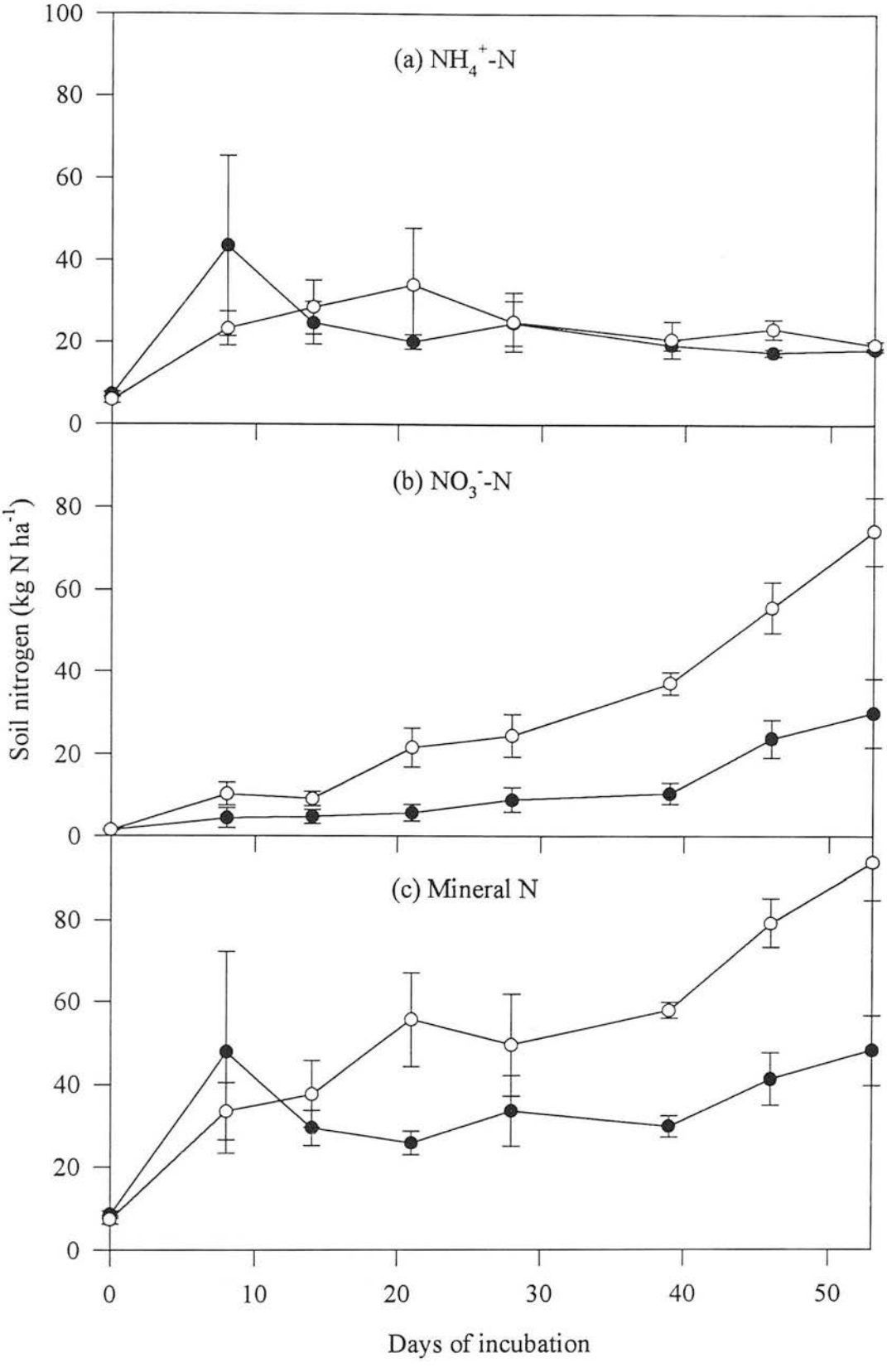

Figure 4.4.1.3 Soil ammonium-, nitrate- and mineral nitrogen concentrations following incorporation of grazed (open symbols) and ungrazed (solid symbols) sward residues (Bars = SE, d.f.=2).

4.4.1.4 Carbon dioxide emissions

Application of acetylene had no significant effect on CO_2 evolution, in agreement with Aulakh *et al.* (1991a), and so data from the GAS and ACET replicates were bulked for statistical analysis.

On both treatments CO_2 evolution showed a sharp increase after incorporation of the sward material, peaking within a few days and subsequently diminishing (Figure 4.4.1.4b). Increased CO_2 evolution was sustained for longer from the grazed treatment and, after day 0, CO_2 emission rates were significantly higher from the grazed treatment ($p<0.05$) on all bar six occasions. This treatment effect was most apparent during the first 18 days. One way analysis of variance showed that the coefficient of variation was significantly greater on the grazed treatment ($p<0.01$).

4.4.1.5 Nitrous oxide emissions

Acetylene application only significantly affected the N_2O fluxes on the days on which it was applied, despite its effect on mineral N form (section 4.4.1.3).

Nitrous oxide fluxes increased to a peak during the first week following incorporation of the swards (Figure 4.4.1.4c). Emission rates increased sharply between day 0 and day 2 from in both treatments. After ten days, fluxes from the ungrazed treatment decreased whilst fluxes from the grazed treatment remained high until day 33. Two way analysis of variance found fluxes from the grazed treatment to be significantly greater than from the ungrazed treatment from day 13 to 39 ($p<0.05$). When the soil was wetted to a gravimetric moisture content of 0.35, on day 47, the significant difference between treatments returned ($p<0.05$). During the experiment coefficients of variation of N_2O fluxes from the grazed treatment were not quite significantly higher than those from the ungrazed treatment ($0.05<p<0.1$).

T-tests found that cumulative N_2O fluxes on the grazed treatment were significantly greater than those from the ungrazed treatment from day 4 onwards ($p<0.05$). Cumulative N_2O fluxes for the 50 day incubation period are shown in Table 4.4.1.5.

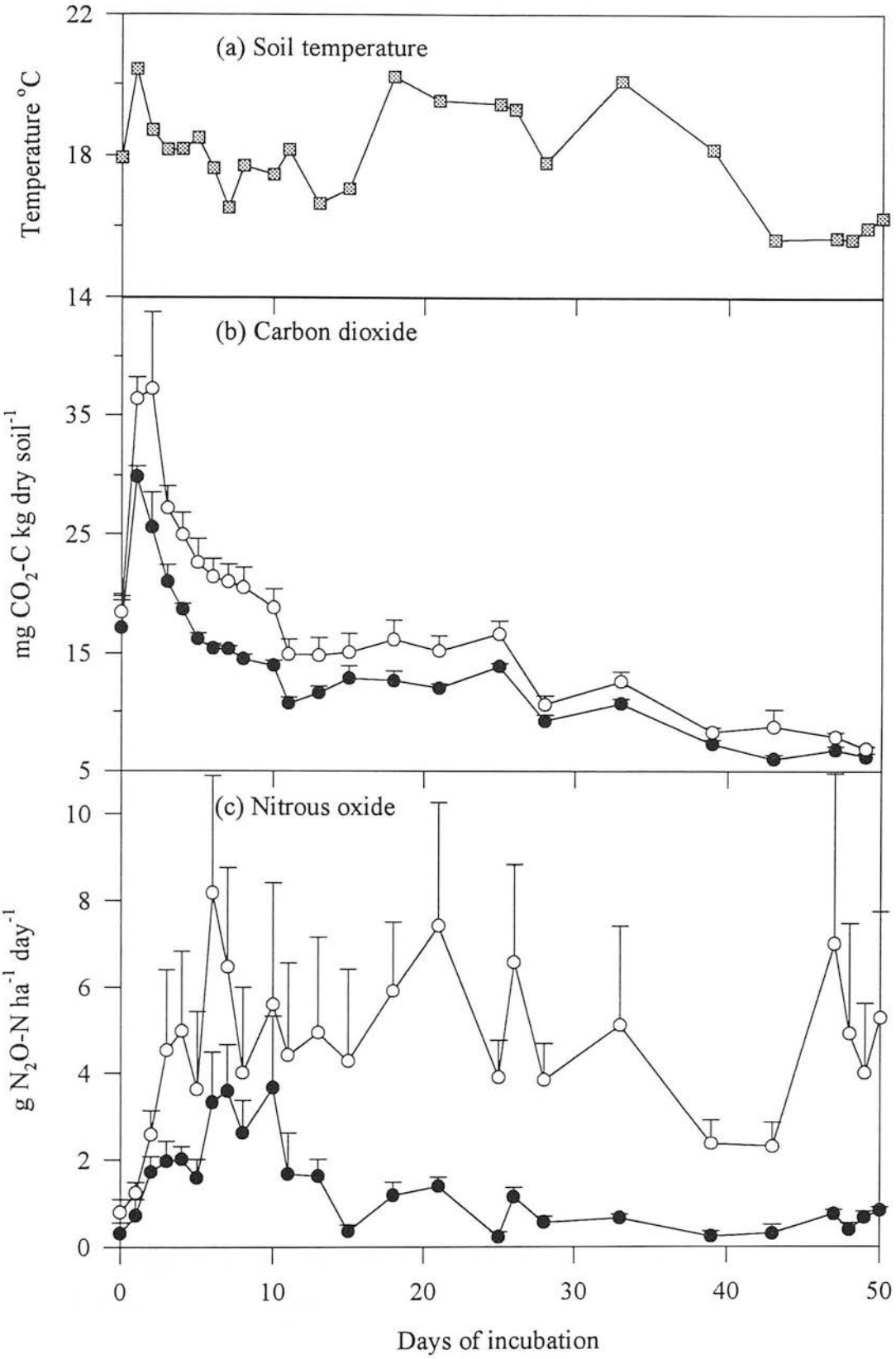

Figure 4.4.1.4 Soil temperature, carbon dioxide and nitrous oxide[a] fluxes (treatment means) following incorporation of grazed (open symbols) and ungrazed (solid symbols) sward residues (Bars = SE, d.f.=5 for (b), 2 for (c)).

[a] Without acetylene.

Table 4.4.1.5 Cumulative gaseous N_2O-N fluxes for the 50 day period following incorporation of treatment residues (Standard errors in parentheses).

Treatment	Cumulative flux (kg N ha^{-1})	
	Grazed	Ungrazed
- acetylene	0.2 (0.1)	0.1 (0.0)
+ acetylene	2.0 (1.3)	0.3 (0.1)

4.4.1.6 Denitrification losses

There was no significant difference between the $N_2 + N_2O$:N_2O ratios from the grazed and ungrazed treatments (Table 4.4.1.6). The $N_2 + N_2O$:N_2O ratios for each of the turf replicates showed no consistent temporal pattern. In order to calculate an approximate total denitrification flux, $N_2 + N_2O$:N_2O ratios were linearly interpolated between denitrification sampling dates for each turf replicate. Using these ratio estimates the total estimated denitrification losses were calculated (Table 4.4.1.5).

Table 4.4.1.6 $N_2 + N_2O$:N_2O ratios (treatment means) during the incubation period (Standard errors in parentheses, d.f.=2).

Day number	Ratio	
	Grazed	Ungrazed
Day 11	10.35 (4.01)	5.18 (1.29)
18	8.92 (3.77)	3.17 (0.04)
28	6.38 (2.53)	6.67 (2.60)
39	6.04 (3.50)	9.12 (3.50)
49	4.55 (1.54)	2.93 (0.33)

4.5 METEOROLOGICAL DATA

4.5.1 Soil temperature and rainfall data

Soil temperature and rainfall data for the period January 1992 to June 1994 are shown in Figure 4.5.1, along with long term average data.

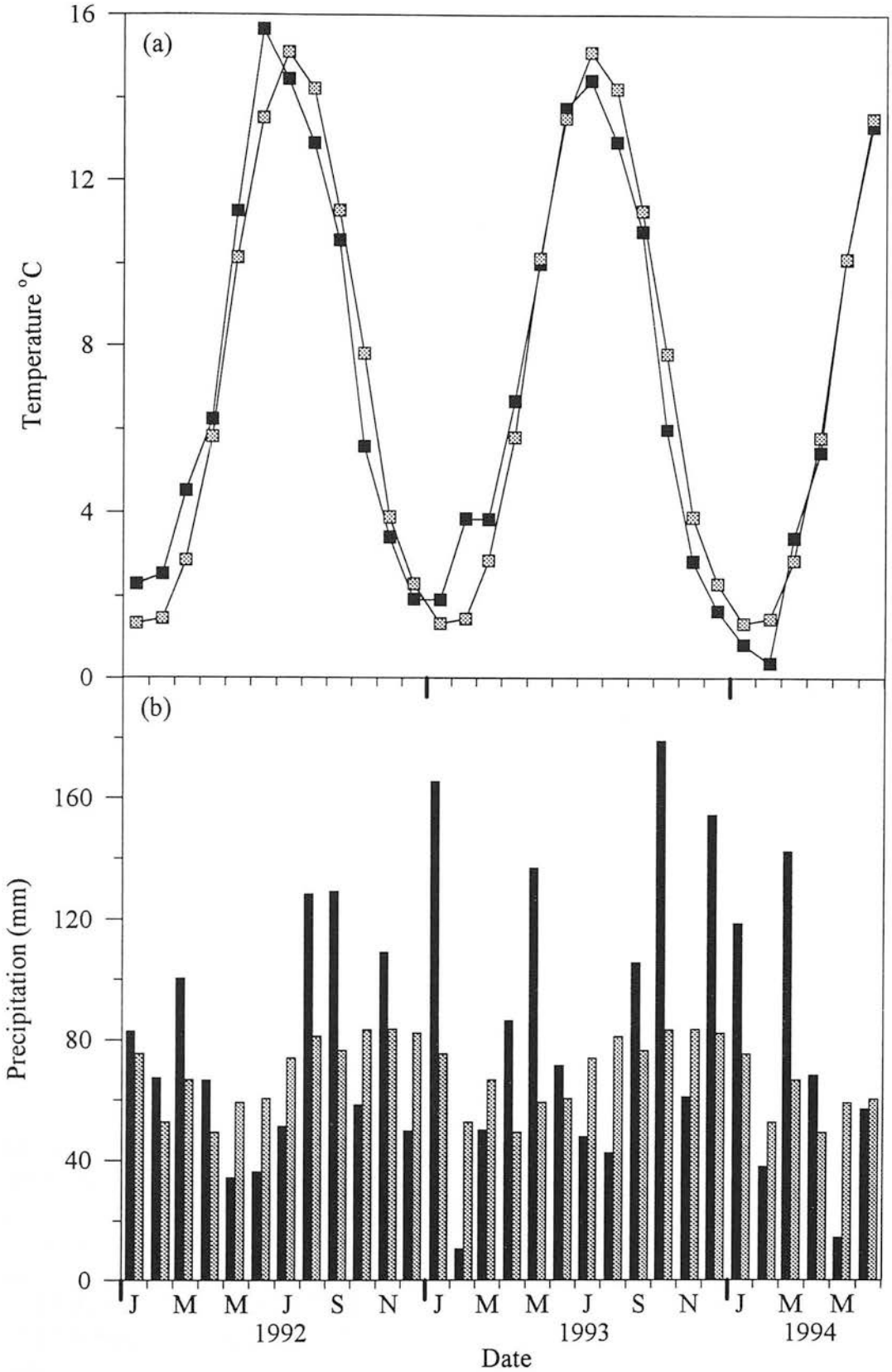

Figure 4.5.1 (a) Monthly average soil temperatures at 10 cm (solid square=1992-1994 data, shaded square=long term average, 1955-1990). (b) Monthly rainfall totals (mm) (solid bars=1992-1994 data, shaded bars=long term average, 1955-1990).

Data obtained from the Bush House Meteorological Station (No. 1643).

In 1992 and 1993, the soil tended to be slightly colder (*ca.* 1 • C) than the long term average between January and June and slightly warmer (*ca.* 1 • C) than the long term average between July and December. In 1994 soil temperatures were similar to the long term average.

The annual and seasonal rainfall totals between January 1992 and June 1994 are given in Table 4.5.1, along with long term average data. In all three years total measured rainfall was greater than the long term average. Between April and June 1992, rainfall was only 81% of the long term average. Between October 1993 and March 1994, rainfall was 56% greater than the long term average.

Table 4.5.1 Annual and seasonal rainfall totals (mm) (a) between January 1992 and June 1994 (b) long term average data.

Time period	Rainfall (mm)			
	1992	1993	1994	Long term average
January-March	250.3	226.1	299.5	194.8
April-June	137.1	295.0	139.8	169.0
July-September	309.0	195.9	-	231.8
October-December	216.9	395.0	-	249.1
Annual total	913.3	1112.0	-	844.7

4.5.2 Drainage data

Calculated over winter drainage for the 1992-3 and 1993-4 drainage periods are given in Table 4.5.2.

Table 4.5.2 Measured cumulative drainage from the Glencorse hydrologically isolated plots for the 1992-3 and 1993-4 drainage periods.

Drainage period	Cumulative drainage (mm)
14 August 1992-30 June 1993	507
17 September 1993-11 March 1994	640

5 DISCUSSION

5.1 The timing of nitrogen release following ploughing

In both years of the Beechgrove trial, soil mineral N concentrations increased immediately after ploughing (Figure 4.1.3.3). Francis *et al.* (1992) also observed rapid increases in mineral N regardless of the timing, or type, of cultivation. Rodgers *et al.* (1985) observed NO_3^--N concentrations of about 100 kg NO_3^--N ha^{-1}, one month after ploughing out an old grass pasture.

The increased NH_4^+-N concentrations following rotavation, observed in both years in the disturbed treatments at Beechgrove, and also in the incubation experiment, probably reflect increased rates of mineralisation from plant residues and SOM. Rees *et al.* (1993) observed high NH_4^+-N concentrations two weeks after incorporation of pea residues, which was presumed to be due to residue mineralisation. An alternative explanation may be short term inhibition of nitrification due to increased soil acidity during intense microbial decomposition of organic matter (Killham, 1994). The subsequent fall, and relative stability, of NH_4^+-N concentrations at Beechgrove was probably due to nitrification rates being greater than ammonification rates.

In 1992, peak soil mineral N concentrations in the fallow treatments, 156 and 248 kg N ha^{-1}, occurred 48 and 58 days after ploughing in the grass and grass-clover residues, respectively. This accounted for 38 and 59% of the total mineral N release observed over eighteen months in the grass and grass-clover resown 1992 treatments (358 and 374 kg N ha^{-1}), respectively (Figure 5.1a). Ludecke and Tham (1971) reported a maximum accumulation of 231 kg N ha^{-1} in the soil profile following the cultivation and fallow of a two-year old grass-clover pasture in mid-summer. The observed peak mineral N level for the grass-clover fallow treatment lies between the estimated first year's N release following ploughing of three- and eight-year old grazed swards receiving 300 kg N ha^{-1} yr^{-1} according to Whitehead *et al.* (1990).

In 1992, N uptake by the resown 1992 swards was much greater than that by the undisturbed swards (Figure 4.1.6.2). This reflects the additional N supplied after ploughing. Between 7 August and 15 October, the apparent recovery of incorporated residue N was 68 and 108 kg N ha^{-1} (26 and 35%) from the grass-clover and grass sward residues, respectively. This positive crop response was greater than that reported by Redman *et al.* (1989), who observed an apparent recovery of only 4%

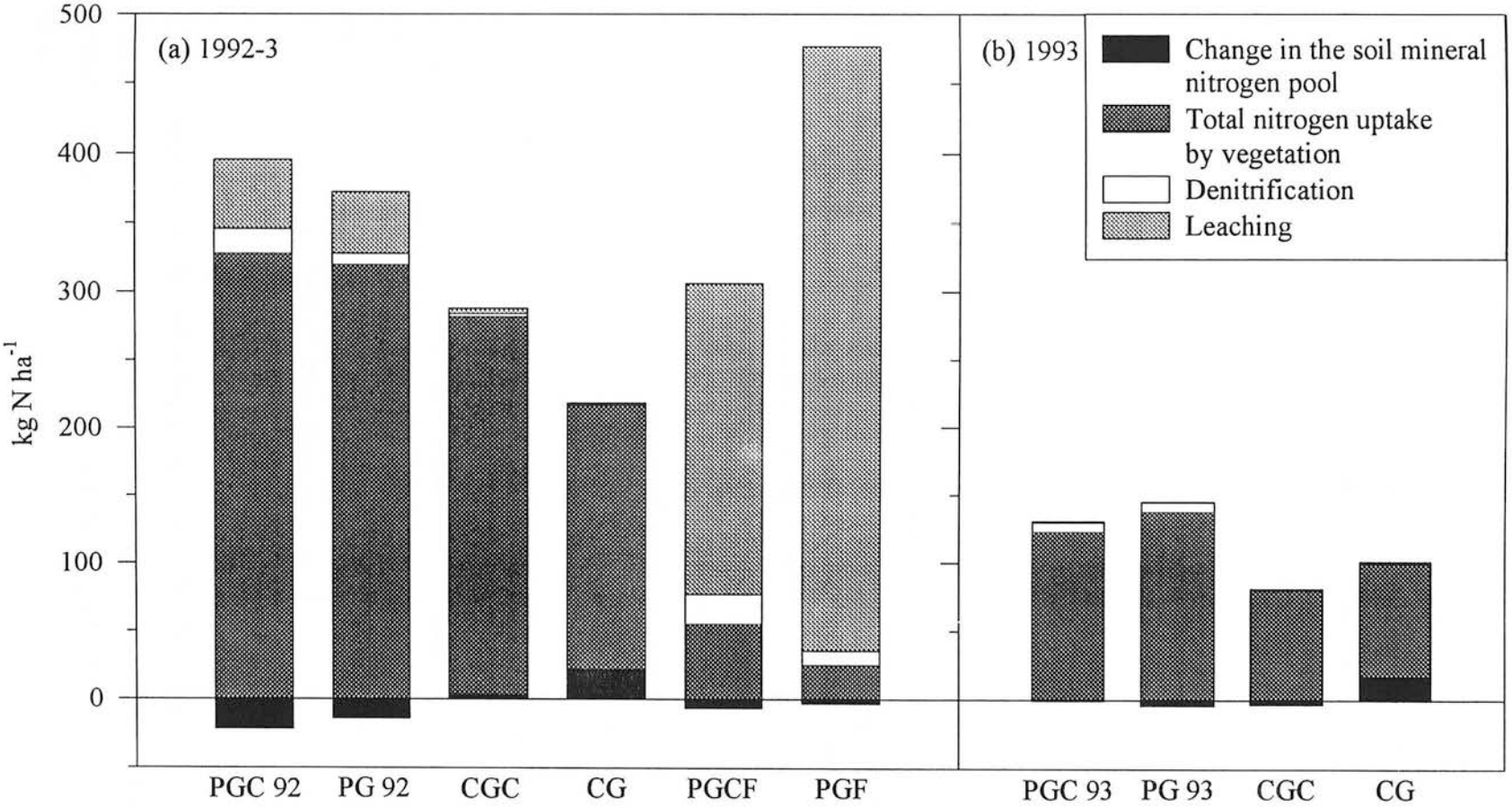

Figure 5.1 Total mineral nitrogen release[a] and the fate of this nitrogen (treatment means) on the Beechgrove trial between (a) 18 June 1992 and 17 December 1993 (b) 11 May 1993 and 17 December 1993.

[a] Shown by the height of the bar except where the change in the mineral nitrogen pool is negative. In these cases, the change in the mineral nitrogen pool should be deducted from the height of the bar.

(equivalent to 13.3 kg N ha^{-1}) in the first crop of winter barley following the autumn incorporation of 335 kg N ha^{-1} of forage pea residues. However, on the Beechgrove trial, N release from SOM cannot be distinguished from that from residues, and is likely to have been much higher than in the trial of Redman *et al.* (1989). This would result in an overestimate of actual plant residue N recovery (section 5.3). Ladd *et al.* (1981b), using ^{15}N-labelled legume residues, reported crop recoveries of 10.9-17.3% fifteen months after legume incorporation, although in a later study 27.8% of legume N was recovered (Ladd *et al.*, 1983).

In 1993, the resown 1993 swards did not overcome the loss of productivity incurred during their establishment, and appeared unlikely to do so, reflecting the slower N release following ploughing in 1993.

In the second cropping season after incorporation, apparent N contributions from residues were 17 and 21% (19 and 25 kg N ha^{-1}) greater than the release from SOM on the undisturbed grass-clover and grass swards, respectively (Figure 4.1.6.2). This greater release occurred mainly in the early part of the growing season and was also shown by the increased mineral N concentrations in the fallow treatments between 21 January and 8 April 1993 (Figure 4.1.3.3b). Francis *et al.* (1992) reported continued mineral N release in the spring following cultivation of grass-clover swards, the amounts depending on the timing of cultivation the previous autumn. The apparent N contributions in the second cropping season were 7 and 8% from the grass-clover and grass sward residues, respectively. These are higher than recoveries of ^{15}N-labelled residues by second crops reported by other workers (Ladd *et al.*, 1983; Ladd and Amato, 1986; Harris and Hesterman, 1990) and, as in the first year, may reflect the additional release from SOM. Nitrogen uptake by the undisturbed swards was actually greater than the uptake by the resown 1992 swards on the last two cutting dates in 1993 (August 9 and October 15). The sharp fall in the N content of resown 1992 herbage after 30 March 1993 (Figure 4.1.6.1) confirms that N release from residues was not sufficient to meet grass crop demand throughout the second year after ploughing, in agreement with Hopkins *et al.* (1990).

The total mineral N released in the eighteen months following ploughing (*ca.* 370 kg N ha^{-1}) was higher than the release over two and a half years, 275 kg N ha^{-1}, estimated by Lloyd (1992) following autumn ploughing of an intensively grazed sward.

5.2 Differences in mineral nitrogen release following ploughing between the two seasons

The principal reason for the difference in residue inputs between 1992 and 1993 was the change to a cutting regime, removing the cycling of N by grazing animals and thereby decreasing the potential supply from SOM. The grass N contents and sward clover contents were lower in 1993, but these factors were of secondary importance.

The grazing animal can have a major effect on N availability following ploughing out of swards because:
a) grazing animals excrete 75-95% of the ingested N (Whitehead, 1970a);
b) excreta returns large amounts of readily mineralisable N to the soil;
c) much of the N returned in excreta is reutilised by the growing sward.

Assuming the grass growing season begins in March, there was about 100 days growth prior to ploughing in 1992. Assuming the dry matter production rate prior to ploughing was approximately double that measured between 7 August and 15 October and, given that N uptake by the undisturbed grass and grass-clover swards during this latter period was 50 and 100 kg N ha^{-1}, respectively (Figure 4.1.6.2) (Frame, 1992d), the potential for N recycling through animals was considerable.

Urine-affected patches receive an application rate equivalent to 150-500 kg N ha^{-1} (Doak, 1952; Sherlock and Goh, 1984; Thomas *et al.*, 1988) and 30-40% of the pasture area may be affected each year (Saunders, 1984). The N content of faeces of sheep fed on grass is usually in the range 2-3% (Whitehead, 1970a), although higher values have been reported (Parsons *et al.*, 1991). Assuming that an average defaecation covers 0.01 m^2, weighing 40 g DM (Frame, 1971; Whitehead, 1995) and has an N content of 3%, each defaecation is equivalent to an application rate of 1200 kg N ha^{-1}.

Sheep urine and cattle dung have been shown to increase soil mineral N for over a month (MacDiarmid and Watkin, 1972; Thomas *et al.*, 1988; Haynes and Williams, 1992). Substantial plant uptake of urinary N has been observed for 3-4 months after application (Fraser *et al.*, 1994), resulting in higher grass dry matter production and N contents in urine-affected than unaffected areas (Cuttle and Bourne, 1993). Much of the N in excreta, particularly that deposited early in the growing season, would therefore have been reutilised by the growing swards. Excretal returns are also

subject to considerable losses of N (Whitehead and Bristow, 1990; Cuttle and Bourne, 1993; Fraser *et al.*, 1994), further decreasing the amount of N returned to the SOM. Nonetheless, the return of N in excreta is a considerable source of readily mineralisable N which was not available in 1993. Immobilisation of up to 20% of urine N has been reported (Whitehead and Bristow, 1990; Fraser *et al.*, 1994) and the release of this N would add to the more immediate N supply from urine.

The effects of excretal returns were detected in the mineral N patterns on the undisturbed treatments at the beginning of the trial (Figure 4.1.3.3d). Under both sward types, the skewness of NO_3^--N concentration distribution, and coefficients of variation, were greater during this initial period than during the latter stages of the trial, indicative of grazing effects (Cuttle, 1992). The greater frequency of samples with high NO_3^--N concentrations in the grass-clover sward suggests that the residual effects of grazing were more prominent on the grass-clover sward (White *et al.*, 1987). Whilst there is no direct evidence of higher stocking densities on the grass-clover sward, prior to fencing in 1992, there was visual evidence of greater compaction on grass-clover plots. Cuttle *et al.* (1992) observed camping areas around gates, fence corners and water troughs. The grass-clover plots were adjacent to all three of these potential camping features. The additional excretal returns in camping areas of grazed pastures have been reported to lead to higher soil organic C and N concentrations than in non-camp areas (Nguyen and Goh, 1992). Higher stocking densities could partly explain the greater N release following ploughing from the grass-clover sward.

The change in management to a cut, unfertilised regime represents a change to a lower N input system, where N fixation by clover is the sole external source of N. Consequently, the soil organic N pool would be expected to move towards a lower equilibrium level, following an asymptotic pattern (Hoogerkamp, 1973; Hassink and Neeteson, 1991). Hassink and Neeteson (1991) reported that the accumulation of soil N was 143-164 kg N ha^{-1} yr^{-1} greater under grazing than cutting, and the C:N ratio of the SOM under the grazed sward was lower. Clement and Williams (1967) reported a smaller difference in accumulation of 50 kg N ha^{-1} yr^{-1}.

During the period of cutting, without fertilisation, swards would have used N derived from mineralisation of readily available SOM (Whitehead, 1984, 1986). With the harvested grass and excretal returns being completely removed, this pool would not be replenished, decreasing the potential N release upon ploughing (Greenland, 1971;

Murata *et al.*, 1995). Williams and Clement (1966) reported that the amount of mineralisable N was 44% greater in frequently grazed than cut swards. Whitehead *et al.* (1990) estimated that a three-year grazed sward, receiving 300 kg N ha^{-1}, would release 66% more N in the first year after ploughing than an equivalent cut sward. The same authors also noted that fertilisation in the year prior to ploughing can significantly influence N release following the ploughing of swards. This is in agreement with Webster and Dowdell (1986), who found that wheat yield, and N uptake, were 50% lower following the ploughing out of unfertilised compared to fertilised grass.

These management effects on soil N supply were shown by the undisturbed treatments N uptake. Between 9 August and 15 October 1993 N uptake by the grass-clover and grass swards was 11.9 and 17.6 kg N ha^{-1}, respectively, only 14 and 34% of uptake over the equivalent period during 1992. Soil temperatures were similar during these two periods, and whilst in 1992 there was higher rainfall during August (1992: 4.1 mm day^{-1}, 1993: 1.4 mm day^{-1}), and to a lesser extent in September, soil gravimetric moisture contents were slightly higher in 1993 (Figure 4.1.4.2). When supplies of soil water are adequate, N is most commonly the key limiting factor for crop production (Goh and Haynes, 1986). The N contents of the harvested fraction of the undisturbed swards were generally within the range of 2-4% N reported by Frame (1992b), but during the later stages of the trial N contents were considerably lower than this, implying that plants were N deficient (Figure 4.1.6.1).

In the incubation experiment, where the effects of herbage quality and climatic conditions were removed, N release from the cut sward residues was 53% lower than that from the grazed sward residues (Figure 4.4.1.3). Translated to the field results, this would account for over half of the difference in release observed between years.

In 1993, the C:N ratios of grass residues (*ca.* 22) were higher than in 1992 (13 and 19 in the grass and grass-clover swards, respectively) (Tables 4.1.2.1 and 4.1.2.2). Whilst this may have slowed net N release, the grass tops are unlikely to have caused any net immobilisation. Due to different sampling techniques in the two years, the N input from grass tops is not directly comparable. However, it seems likely that it was slightly lower in 1993, due to a lower dry matter input and the lower grass N content.

Clement and Williams (1967) reported that, over three years, organic N accumulation under grass-clover swards was 54% higher than under grass swards, equivalent to

40 kg N ha^{-1} yr^{-1}. This may also explain part of the greater release from the grass-clover than the grass sward residues in 1992 on the Beechgrove trial. However, in the Cowloan field trial there was no significant difference in N uptake by spring barley following the 5-year old grass and grass-clover swards (Appendix 7).

Lindén and Wallgren (1992), in an extensive Swedish trial, observed a much greater difference in release from grass-clover and grass swards. This was probably due to the greater importance of N release from plant residues in their trial. Lindén and Wallgren (1992) used two-year old cut leys and therefore the accumulation of SOM may have been less than under the older, grazed swards at Beechgrove. In addition, the difference in clover contents of the two sward types used in their trial was much larger than at Beechgrove (section 5.3).

5.3 The influence of plant residue quality on the release of nitrogen following ploughing

Whitehead *et al.* (1990) stressed the important contribution of unharvested plant fractions and soil macro organic matter to mineral N release following ploughing of grassland. Plant tops are the most readily decomposable fraction of sward residues. In the present study, clover tops had the highest N content of all the ploughed in residues, with a C:N ratio of *ca.* 10:1, and were therefore expected to release N very quickly. Müller and Sundman (1988) reported the rapid release of N from clover residues even at low soil temperatures. Whilst the clover content at Beechgrove was higher in the grass-clover sward in 1992, it was only equivalent to 3 kg N ha^{-1} more N input from tops, and would therefore only make a minor contribution to N release from sward residues. Clover roots may also be expected to have a higher N content than grass roots (Whitehead, 1970b), adding to the potential N release from the grass-clover sward residues. However, unless clover comprises a large proportion of the sward, its effects are likely to be small because the weight of clover roots is low (Young, 1958; Whitehead *et al.*, 1990). The temporal changes in sward clover contents during the trial (Table 4.1.6) are unlikely to have had a significant influence on the difference in N release in the two seasons.

At Glencorse, the effects of clover residues on mineral N release were greater than at Beechgrove (Figures 4.2.2.4 and 4.1.3.3). This was due to the greater N input from clover and the greater contrast in quality between the grass and the clover residues at Glencorse (sections 4.1.2, 4.1.6 and 4.2.1). The N input from clover tops on the

grass-clover sward was 47.0 kg N ha^{-1}, whilst the respective input at Beechgrove in 1992 was only 9.2 kg N ha^{-1}. The C:N ratio of grass residues was 45-56, higher than at Beechgrove in 1992 (section 5.2). According to reported equivalence points (Harmsen and van Schreven, 1956; Harris, 1988 - section 2.2.2.1), the grass residues at Beechgrove probably released N, whilst the grass at Glencorse may have caused immobilisation. In addition, the grass ploughed in at Glencorse had grown to a considerable height. Advancing maturity is usually accompanied by an increase in the proportion of cell wall contents (i.e. cellulose, hemicellulose and lignin) in relation to cell contents (i.e. water soluble carbohydrates) (Frame, 1992e). This would accentuate the difference between the N release patterns of the grass and the clover residues at Glencorse.

Hoogerkamp (1973) reported that SOM accumulation started almost immediately after grassland establishment and seemed to be faster on heavier soils. The significantly higher N uptake by spring barley following a 1-year old grass-clover ley than under continuous arable cultivation ($p<0.01$) would appear to support this argument (Appendix 7). Nitrogen release from SOM was therefore also expected to be higher in the PG treatment than the treatments previously cropped with barley (AN and CN). However, mineral N concentrations did not follow this expected pattern, and were lower in the PG treatment than the AN and CN treatments, suggesting that grass residues immobilised considerably amounts of N. Vinten *et al.* (1996) reported that 27% of the ^{15}N-fertiliser applied to the PG treatment was immobilised, supporting this argument.

The relatively stable mineral N concentrations in the AN and CN treatments prior to fertilisation were probably due to the balance between N release from SOM and any immobilisation caused by the decomposition of the previous barley crops residues. Barley residues would have been decomposing for several months prior to the beginning of soil N measurements, and thus anticipated rates of decomposition (and immobilisation) would be small (Amato *et al.*, 1987). It would therefore appear that N release from SOM in the AN and CN treatments was very small.

Ladd *et al.* (1981b) reported that 71.9-77.7% of ^{15}N-labelled legume residues remained as organic N after 15 months decomposition, whilst Nicolardot *et al.* (1995) found 41.9% of ^{15}N-labelled residue from a ryegrass catch crop was released into inorganic form after 5 months. Given the plant tops residue N input (Table

4.1.2.1), even using the upper release estimate, the supply of N from other sources was clearly substantial.

Plant roots provided a substantial potential N supply. However, the N contents of these plant fractions were much lower (C:N ratio *ca.* 30), and may be associated with net immobilisation of soil mineral N during initial stages of decomposition (Power, 1968; Steele and Vallis, 1988). The chemical composition of roots also makes them generally less decomposable than tops. Müller *et al.* (1988) reported higher lignin and cellulose concentrations, and lower soluble substances, in roots of timothy grass and red and white clover. In every species investigated, the percentage of N released by roots was lower than the average for the whole plant (Müller *et al.*, 1988), and other workers report similar findings (Amato *et al.*, 1987; Harris and Hesterman, 1990).

Results from field and incubation experiments at Beechgrove give no direct indication of the relative importance of plant residues and SOM in mineral N release. Francis *et al.* (1992), using ^{14}C-labelled ryegrass, found that about 40% and 55% of residues had decomposed two and five months, respectively, after incorporation. Amato *et al.* (1987) observed that the amount of N released after eight weeks from legume residues (C:N 12-14) was about 30-40%. Assuming the lower release rate (30%), plant residues accounted for 78 and 94 kg N ha^{-1} of the N released in the eight weeks after ploughing the grass-clover and grass swards, respectively. This was equivalent to 36% and 70% of the net mineral N released between ploughing and 7 August and 28 July in the PGCF and PGF treatments, respectively. It is concluded, therefore, that a substantial part of the release of mineral N after incorporation was from SOM. This could also explain the higher N uptake by spring barley following a 1-year old than a 5-year old grass-clover ley on the Cowloan trial (Appendix 7).

Nitrogen uptake by the undisturbed treatments between 7 August and 16 September was *ca.* 70 and 27 kg N ha^{-1} on the grass-clover and grass swards, respectively. Assuming the daily N uptake rate between ploughing (10 June) and the occurrence of peak mineral N concentrations in the fallow treatments (28 July and 7 August) was *ca.* 30% higher than that between 7 August and 16 September, it is estimated that *ca.* 43 and 133 kg N ha^{-1} was released from SOM between ploughing and 28 July and 7

August in the grass and grass-clover blocks, respectively[4]. The greater N release from SOM in the grass-clover block compensates for the lower calculated N release from grass-clover than grass sward residues. The effects of soil disturbance would further stimulate the release of soil organic N (Craswell and Waring, 1972).

5.4 Gaseous losses of nitrogen following sward incorporation

In all trials the period of greatest N_2O emissions occurred shortly after incorporation of sward residues (Figures 4.1.4.1, 4.2.3 and 4.4.1.4c). The incorporation of fresh plant residues provides a readily available C supply and causes rapid increases in microbial activity (Figure 4.4.1.4b; Cerri and Jenkinson, 1981; Redman *et al.*, 1989; Aulakh *et al.*, 1991b, 1991c; Janzen and McGinn, 1991; McKenney *et al.*, 1993). Webster and Dowdell (1986) reported enhanced N_2O emissions following incorporation of a fertilised grass sward, to values about five times those from undisturbed grass, for two months after incorporation. Emissions reported by these authors were lower than at Beechgrove which may have been due to the later ploughing date, early October, and consequently lower soil temperatures.

Nitrous oxide flux patterns on the two trials suggest that rotavation, rather than ploughing, initiates the increase in emissions. At Beechgrove on 18 May 1993 N_2O emissions from the EP plot and resown 1993 treatments were very low, despite over 90 mm rainfall in the previous five days. Since soil NO_3^--N concentration in the EP plot was not significantly lower than in the PGCF treatment, from which emissions did increase, it seems likely that microbial activity was insufficient to create anaerobic conditions. Following rotavation, much lower rainfall (30 May (8.5 mm) and 1 June (11.5 mm)) led to increased N_2O fluxes on 2 June from the resown 1993 treatments, and to a lesser extent from the EP plot. The continued rise in fluxes from the rotavated treatments, despite no further rain, was due principally to a 10°C increase in soil temperatures, further stimulating microbial activity and anaerobiosis.

At Glencorse, six days after ploughing out of swards, N_2O fluxes were less than 5 g N_2O-N ha^{-1} day^{-1} despite 25 mm of rainfall since ploughing. Since soil NO_3^--N concentrations in the PG and PC treatments were *ca.* 10 kg NO_3^--N ha^{-1}, sufficient

[4] The herbage N uptake data taken between 7 August and 16 September 1992 includes some N fixation by clover in both swards. Therefore these figures are overestimates of N release from SOM. However, on 7 August sward clover contents were only *ca.* 3 and 5% of total DM on the grass and grass-clover swards, respectively (Table 4.1.6). Given the small difference in sward clover content, N fixation can only account for a small fraction of the difference in N uptake on the two swards.

for large denitrification fluxes (>0.15 kg N ha^{-1} day^{-1}; Ryden, 1986), it appears that microbial activity was insufficient to produce anaerobic conditions. Following rotavation, despite negligible rainfall and lower soil moisture contents, fluxes from the PC treatment increased.

The greater N_2O emissions following rotavation, as opposed to ploughing, are probably due to more thorough incorporation of residues, increasing microbial oxygen demand and anaerobic microsite development. Similar effects have been observed following the incorporation of grass and grass-clover swards at another site on the Bush Estate, Midlothian (Baggs *et al.*, 1996).

The importance of anaerobic hotspots was emphasised in the incubation experiment where, given the low WFPS and high porosity of the soil, anaerobic conditions were unlikely to have been due to saturated soil. Furthermore, peak N_2O emissions lag behind those for CO_2, as was observed by Christensen and Tiedje (1988), emphasising the link between microbial activity, oxygen depleted zones and emissions (Figure 4.4.1.4).

5.4.1 Beechgrove

In 1992 the soil was much drier than in 1993 (Figure 4.1.4.2). In the twelve days between ploughing and rotavation there was only 0.3 mm rainfall in 1992, whilst over the comparable period in 1993 there was 105.3 mm rainfall. Many workers have reported a burst of microbial activity, associated with an expansion in microbial populations, following the rewetting of a dried soil (Birch and Friend, 1956; Stevenson, 1956; Campbell and Biederbeck, 1982). Drying kills part of the soil microbial population, which is larger under grassland than cultivated soils (DeLuca and Keeney, 1994), and increases the availability of degradable organic C (Ayanaba *et al.*, 1976; Patten *et al.*, 1980). Rolston and Liss (1989) found that air-drying of soils for 48 hours more than doubled the water soluble organic carbon content of a fine, silty soil. In 1992 the rainfall following rotavation, falling on very dry soil, may have produced a burst of microbial activity due to rewetting and, simultaneously initiated the rapid decomposition of the incorporated residues. In 1993 soil moisture was unlikely to have limited microbial activity and decomposition would have started immediately upon incorporation. Consequently, in 1993 the burst of microbial activity following residue incorporation may have been much smaller.

The lower soil temperatures prior to and following ploughing in 1993 would also have lessened microbial activity. Mean soil temperatures were 6.5°C higher during the three weeks after ploughing in 1992. Reported Q_{10} values for N_2O emission over the temperature range experienced at Beechgrove (4-18°C) range between 0.9 and 14 (Nõmmik, 1956; Keeney *et al.*, 1979; Smith *et al.*, 1995).

Sward management between ploughing in 1992 and 1993 affected gaseous N losses following sward incorporation. An incubation experiment showed significantly higher water soluble organic carbon, microbial activity and N_2O emissions from grazed than cut sward residues following incorporation (Figure 4.4.1.4, Table 4.4.1.2). Sheep dung contains more C and less readily available N than urine (Whitehead, 1986; Latinga *et al.*, 1987) and takes several months to break down completely (Castle and MacDaid, 1972). It therefore seems likely that dung has a greater influence on soil microbial activity and potential 'hotspot' creation. The gradual decomposition of dung in combination with the more readily available SOM (section 5.2), may have sustained the higher microbial activity from the incorporated grazed sward residues. Dung, being non-uniformly distributed over the sward, may also be the cause of the greater spatial variability of CO_2 and N_2O emission rates from this treatment.

Residual effects of grazing may also explain the greater gaseous N emissions from the grass-clover than from the grass block at Beechgrove in 1992 (section 5.2). However, peak denitrification fluxes from the undisturbed swards were lower than those reported from excreta-affected pasture by other workers (Ryden, 1986; Colbourn, 1992).

The change in management also altered the herbage residue inputs in the two seasons. In 1993, the clover content of the grass-clover sward was lower than in 1992, whilst that of the grass sward had increased (Table 4.1.6). The effect of clover residues was evident in N_2O emission patterns in the Glencorse field trial (section 5.4.2) but the influence of clover was much smaller at Beechgrove (section 5.3).

In 1993 the dry matter inputs were smaller than in 1992, primarily due to lower inputs from 'tops' and '0-4 cm' fractions. Addition of the 'tops' and '0-4 cm' fractions shows that inputs were *ca.* 2400 and 3300 kg DM ha^{-1} lower in 1993 from the grass-clover and grass swards, respectively. This would be particularly important if the input from plant tops was greater in 1992, since plant tops generally provide more readily available C sources than roots (Jenkinson, 1977a; Frankenberger and

Abdelmagid, 1985). Jenkinson (1977a) showed that the rate of CO_2 emission was roughly proportional to the amount of ryegrass tops added to soil. The greater addition of plant tops in 1992 would therefore have been likely to produce more anaerobic zones. McKenney *et al.* (1993) observed a 70% increase in N_2O emissions with a doubling of the residue quantity added. The greater input of plant tops from the grass-clover than the grass sward in 1992 may have contributed to the higher emissions from the grass-clover treatments.

In conclusion, no one factor can completely account for the difference in gaseous N losses between the two years. Rather, it appears to have been due to the different soil moisture and temperature conditions, combined with changes in residue inputs, making conditions for gaseous N losses more optimal, at critical times.

The lower fluxes from the undisturbed treatments reflects the lower C and NO_3^--N availability. Fluxes after 17 July were similar to those from unfertilised grass at an adjacent field site (Clayton *et al.*, 1996) but below the 6-25 g N_2O-N ha^{-1} day^{-1} reported by Denmead *et al.* (1979) from moist soil under unfertilised swards.

In 1992 flux measurements did not begin until ten days after rotavation, 22 days after ploughing. Whilst this represents a considerable gap in the data set, from the discussion above it seems likely that during much of this period fluxes would have been low. However, fluxes may have increased following rotavation, despite the dry conditions, due to the intense microbial activity. The rainfall on 29 (10.3mm) and 30 June (17.3 mm) may also have resulted in fluxes greater than those detected two days later when measurements began, since both nitrifying and denitrifying bacteria are active within minutes of the wetting of dry soil (Davidson, 1992).

5.4.2 Glencorse

At Glencorse, the effect of the various residue treatments had a marked effect on observed N_2O emissions (Figure 4.2.3). The higher emissions from the grass-clover treatment, particularly prior to fertiliser application, were probably due to the greater clover residue input. The N-rich clover residues probably provide more readily decomposable C than the low-N grass residues. Clover would therefore make more demands on the oxygen supply and may have created larger anaerobic zones than grass residues. McKenney *et al.* (1993) observed that N_2O emissions were 2-3 times higher from legume than ryegrass residues. deCatanzaro and Beauchamp (1985)

found that alfalfa caused more N_2O emissions than straw in the first three days after incorporation. These authors suggested that this was due to the higher soluble water extractable C from alfalfa, as well as to its lower lignin and cellulose content.

The position of C sources in the soil profile relative to NO_3^--N will influence the rate of denitrification (Aulakh *et al.*, 1992). The grass residues are likely to have caused immobilisation of existing soil mineral N (section 5.3), thereby competing with the denitrifiers for substrate in the surrounding soil. In contrast, clover decomposition is likely to result in net mineralisation (section 5.3), and subsequent production of NO_3^--N. Therefore, for denitrification to occur in any anaerobic microsites caused by clover residues would not require diffusion of NO_3^--N to the anaerobic zone. Any inhibition of denitrification in the PG treatment due to lack of NO_3^--N would probably have been removed following fertiliser application. Aulakh *et al.* (1984) showed that incorporation of wheat residues with 100 kg $(NH_4)_2SO_4$ ha^{-1} resulted in a doubling of denitrification losses compared to those from wheat residues alone.

After the initial burst of activity, whilst the stimulation of microbial activity due to residue addition remained important, emissions were increasingly dependent on rainfall to create anaerobic conditions. Bergos (1992) suggested that a decline of available C meant that oxygen demand was insufficient to cause anaerobic conditions. This is supported by the significant decline in water soluble organic carbon during the incubation experiment (Table 4.4.1.2).

On 13 August 1993 emissions increased from the PC and PG treatments whilst fluxes from the CN treatment remained low. Moisture levels in the CN treatment, following the 12.4 mm rainfall since the previous sampling date, were higher than those observed in the PC treatment on 27 July, when the latter treatment produced significant fluxes. This suggests that rainfall alone was not sufficient to create anaerobic conditions, and it was the greater microbial activity in the PC and PG treatments which explains the different treatment fluxes.

The major period of N_2O fluxes from all treatments was probably due to soil saturation. The 12.2 mm of rainfall on 8 September wetted the top few centimetres of soil. Between sampling on 9 and 16 September, over 75 mm of rain fell, concentrated in two major events on 13 (22 mm) and 14 September (28 mm). This would have saturated the surface soil to an even greater depth, producing a large increase in fluxes in all treatments. The sharpest increases and highest fluxes were observed from

those treatments with large amounts of residues incorporated. This suggests that whilst the rainfall was able to induce anaerobic zones in the soil, the additional oxygen demand due to microbial decomposition of residues in treatments PG and PC increased and sustained these anaerobic zones, causing higher fluxes. In contrast, Goulding *et al.* (1993) found no significant difference between denitrification losses of applied fertiliser from winter wheat following a three-year grass-clover ley or continuous arable cropping. These authors suggested that the improved structure and porosity outweighed the effects of greater C and N content following the ley.

The lack of available C substrate in the CN and AN treatments was reflected by the continued low fluxes on 8 October, despite over 100 mm rainfall in the three days prior to sampling. In contrast, notable increases in fluxes were seen from plots 1, 5 and 8.

The fluxes recorded from the PG and PC treatments were much higher than those reported by Arah *et al.* (1991) at the same site following autumn incorporation of 335 kg N ha^{-1} of forage pea residues (3.1% N). This is probably explained by the lower soil temperatures encountered in the trial of Arah *et al.* (1991).

Reported N_2O yields (Appendix 5) show slightly higher median values for ammonium nitrate fertiliser than nitrate fertilisers (Bouwman, 1990; Eichner, 1990), but the range of reported values is very similar. Nitrous oxide yields at Glencorse, calculated without unfertilised control treatments, and therefore overestimates, were also slightly higher from the ammonium nitrate fertiliser, with yields from both fertiliser types falling in the middle of the reported ranges.

During the incubation experiment, the $N_2+N_2O:N_2O$ ratios varied considerably between turf replicate sites (section 4.4.1.6), and sampling dates. Arah and Smith (1990) found that N_2O accounted for between 5 and 100% of the gaseous products of denitrification on the same soil. The calculated denitrification fluxes reported in sections 4.1.4, 4.2.3 and 4.4.1.6 must be interpreted with some caution since $N_2:N_2O$ ratios were not determined on every sampling occasion.

5.5 Nitrate-N leaching losses following sward incorporation

At the onset of drainage in 1992, soil water NO_3^--N concentrations in all ploughed out treatments at Beechgrove were above the EU maximum acceptable concentration

of 11.3 μg NO_3^--N ml^{-1} (Figure 4.1.5.1.1). At Cowloan leachate concentrations were always below the EU limit, probably due to the earlier ploughing date, strong regrowth following the barley crop and previous cutting of swards. Lloyd (1992) reported mean drainage water concentrations ranging from 4 to 47 μg NO_3^--N ml^{-1} over the first winter following autumn ploughing of grass swards, with one heavily manured site having a mean concentration of 269 μg NO_3^--N ml^{-1}. Bergström (1987) observed lysimeter drainage concentrations of 10-20 μg NO_3^--N ml^{-1} following autumn ploughing of a four-year old fertilised grass sward.

Nitrate-N concentrations showed a smooth temporal pattern for individual cups (Appendix 4). There was large intra- and inter-plot variability, with peak concentrations often occurring at different times, in agreement with the findings of other workers (van de Pol, 1977; Lord and Shepherd, 1993). Average coefficients of variation (CV's) within plots ranged from 26% (PGCF3) to 97% (PGC92 2), the higher CV's in PGC92 2 perhaps reflecting the increased variability caused during establishment of the resown sward. Both plot and treatment CV's were frequently up to 170% during the trial, greater than those reported by Lord and Shepherd (1993) for grazed swards (50-100%) or following cereals (35-55%).

The lower soil NO_3^--N:NH_4^+-N ratio observed in the undisturbed treatments than the fallow treatments, at both Glencorse and Beechgrove (Figures 4.2.2.4, 4.1.3.1 and 4.1.3.2), is in agreement with the findings of other workers (Woodmansee *et al.*, 1981; Bergström, 1986; Mallarino and Wedin, 1990; Parsons *et al.*, 1991). Soil NO_3^--N concentrations in unfertilised grass swards are usually small because of exhaustive uptake of N and plant-induced immobilisation reactions (Legg and Meisenger, 1982). In 1993, the fully established resown 1992 swards showed a similar distribution of soil mineral N forms to the undisturbed swards. The lack of N in a form available for leaching explains the small NO_3^--N loads from these treatments. Bergström (1987) reported losses ranging from 0.2 to 4.6 kg N ha^{-1} from fertilised grass leys, with concentrations commonly below 0.1 μg NO_3-N ml^{-1}. Webster and Dowdell (1986) reported average losses of 3.8 kg N $ha^{-1}yr^{-1}$ from unfertilised grass on a clay soil.

In the grass block between 10 June 1992 (ploughing) and 6 June 1993, the greater NO_3^--N leaching losses from the fallow than the resown 1992 treatment (207 kg N ha^{-1}) fitted quite well with the greater N uptake by the resown sward (243 kg N ha^{-1}). In the grass-clover block the difference in leaching losses (57 kg N ha^{-1}) was

much less than the difference in vegetation N uptake (228 kg N ha^{-1}). On the grass-clover fallow treatment, potentially leachable soil NO_3^--N reached a peak of 204 kg N ha^{-1} on 7 August (Figure 4.1.3.2). Further NO_3^--N release may have occurred after this peak in 1992, and was seen to occur in early 1993. Even allowing for N uptake by regrowth, leaching losses from the grass-clover fallow treatment appear to be considerably underestimated.

Lloyd (1992) reported leaching losses of 100 kg N ha^{-1} from wheat following spring cultivation of an intensively grazed grass sward. The lower leaching losses from the resown 1992 treatments at Beechgrove (Figure 5.1) may reflect the longer growing season of grass than that of wheat (Powlson, 1993). Ludecke and Tham (1971) reported losses of 169 kg N ha^{-1} following the cultivation and fallow of a two-year old grass-clover pasture in mid-summer, falling between calculated losses for the fallow treatments on the two sward types at Beechgrove. Much higher losses, up to 500 kg N ha^{-1}, have been recorded according to Croll and Hayes (1988). Francis *et al.* (1992) reported lower losses, 40-78 kg N ha^{-1}, when swards were ploughed out later in the season.

Over the second winter estimated leaching loads for the fallow treatments, 197 and 296 kg N ha^{-1} in the grass-clover and grass blocks, respectively, were higher than those calculated for the previous winter (Table 4.1.5.2). Soil NO_3^--N concentrations (0-20 cm) in the fallow treatments prior to the onset of drainage in 1993 were 74 and 80 kg NO_3^--N ha^{-1} on the grass-clover and grass blocks, respectively (Figure 4.1.3.2). Even if it is assumed that the 20-40 cm soil layer contained the same amount of NO_3^--N, the calculated leaching appears to be a considerable overestimate. Lloyd (1992) suggested that continued release of N from readily decomposable SOM could explain the poor correlation between mineral N in the soil profile and calculated leaching losses. At Beechgrove the discrepancies may have been exacerbated by the variable performance of porous cups (section 5.6).

The negligible losses from the resown 1993 treatments (Figure 5.1) reflects the lower N release following ploughing in 1993, and also the earlier sowing date. Webster and Dowdell (1986) observed similar leaching losses, 4 kg N ha^{-1}, following autumn ploughing of an unfertilised sward sown to winter wheat.

5.6 The performance of porous cups in estimating solute leaching

The considerable difference in soil water NO_3^--N concentrations measured using the four sampling techniques at Glencorse (section 4.2.4.3), and the apparent discrepancies between calculated and expected loads at Beechgrove (section 5.5), provokes serious questions regarding the use of porous cups on medium textured or heavy soils.

Results from the Beechgrove and Glencorse field trials generally showed that at the beginning of the drainage season, when NO_3^--N concentrations were highest (regardless of measurement technique), porous cups provided the highest estimate of NO_3^--N concentrations in soil water. In contrast, later on in the drainage season, when NO_3^--N concentrations were much lower, porous cups provided the lowest estimate of NO_3^--N concentrations in soil water (Figure 4.2.4.1).

Whilst both porous cup and soil coring techniques detected the pulse of NO_3^--N moving through the soil, and were generally in qualitative agreement, they appear to have sampled different fractions of the soil solution. Hansen and Harris (1975) state that each pore size group can have its own unique volume of drainable water, ion concentration and drainage rate curve. Consequently, considerable differences in solution chemistry may result if different sampling techniques are not sampling the same pore size fraction.

A possible cause of the higher concentrations detected in pore water than in drains and wells at Glencorse may be the source of the solute in the soil. This may also explain the extent of agreement between sampling techniques on the three plots at Glencorse at the beginning of the drainage season. If mineralisation occurred within soil aggregates, any NO_3^--N subsequently produced (i.e. resident) would be detected by volume-averaged sampling techniques (i.e. soil core analysis) regardless of the dominant soil water flow path. When bypass flow occurred, drains or wells would have sampled water which had had very limited contact with solutes within aggregates during its flow through the soil, resulting in lower solute concentrations. Under plot 8, the higher NO_3^--N concentrations in pore water than in drains or wells, may have been because NO_3^--N was entirely resident, mineralised from grass-clover residues and SOM. The same situation applied to a lesser extent on plot 4, since some of the applied fertiliser became involved in the soil heterotrophic subcycle (sections 2.2.2 and 5.3). Under plot 7, the lower NO_3^--N concentrations in pore

water than in drains or wells may have been because the primary source of N was applied fertiliser (i.e. non-resident), much less of which was immobilised than in plot 4 (Vinten *et al.*, 1996). Magesan *et al.* (1994) also found that when an externally applied solute (Br^-) was sampled, concentrations were lower in soil cores than cups and drainage.

Goulding and Webster (1992) found that, prior to fertiliser application, NO_3^--N concentrations in porous cup samples were higher than in field drainage water, but after fertiliser was applied the situation was reversed. These authors suggested that on this free draining soil, with some structure, porous cups sampled water that was relatively immobile compared with drainage water. If porous cups tended to sample static, or less mobile water at Glencorse, the source of the NO_3^--N (i.e. resident or non-resident) may also explain the higher NO_3^--N concentrations in porous cup samples than drains or wells. Consequently, the greater the proportion of NO_3^--N that was resident (plot 8 > 4 > 7), the greater would have been the disparity between NO_3^--N concentrations in cup samples and drain and well samples (plot 8 > 4 >7) (Figure 4.2.4.1). However, this does not explain why NO_3^--N concentrations in porous cups were higher than in pore water.

de Haan and Bolt (1963) showed that anion exclusion from portions of the soil solution may occur due to electrical repulsion of negatively charged clays. Alberts *et al.* (1977) suggested that the net result is higher concentrations in solution held at lower suctions (such as the -20 kPa sampled by cups) than the average of the soil solution. However, other workers considered anion exclusion to be negligible (van de Pol, 1977; Grossman and Udluft, 1991), particularly at high concentrations when the electric double layer will be thinner (Krupp *et al.*, 1972; van de Pol, 1977).

Webster *et al.* (1993) found lower NO_3^--N concentrations in cores than porous cups when no fertiliser was applied. In contrast, Magesan *et al.* (1994) found that when solutes were resident (NO_3^--N and Cl) concentrations were higher in cores than porous cups, as did Alberts *et al.* (1977). Webster *et al.* (1993) suggested that the differences may be due to different solute concentrations in the very small pores, which would only be sampled by soil coring.

Lloyd (1992) suggested that mineralisation of residues during the winter following ploughing of grass may be considerable. This resident NO_3^--N supply may explain why, in January at Glencorse (Figure 4.2.4.1), and to a lesser extent at Beechgrove,

NO_3^--N concentrations in porewater were higher than concentrations measured using all the other sampling techniques. An alternative explanation may be that soil sampling has poorer sensitivity for NO_3^--N than direct solution sampling, at the same analytical detection limit, because of the need for an extraction stage which results in dilution (Lord and Shepherd, 1993). Webster *et al.* (1993) point out that if soil solution N concentrations are near to the EU limit of 11.3 μg NO_3^--N ml^{-1}, the inorganic N concentration in a soil extract is close to the limits of detection. Estimated pore water concentrations below 5 μg ml^{-1}, which frequently occurred at the end of the drainage season, must be treated with a great deal of scepticism, and are probably overestimates.

Following Br tracer application, the peaks in Br concentrations from all cups on 5 May 1993 (Figure 4.1.5.3) appeared to represent detection of the uniform displacement of the discrete pulse of Br that had diffused into the soil matrix. Approximate calculations of tracer displacement using the equation in Appendix 1.2 showed good agreement with the timing of detected Br peaks in the porous cups at 40 cm depth. However, calculated Br loads up to the week ending October 8 1993 (Table 4.1.5.3) only accounted for *ca.* 14% (40 cm) and 9% (55 cm) of the Br applied. Vegetation was estimated to have taken up *ca.* 46% of the applied Br. A further 6% of applied Br still remained in the 0-20 cm soil layer on 12 October 1993, with concentrations below this depth being the same as control samples. About one third of Br was therefore lost from the soil system without detection.

Germann and Beven (1981) stated that macropores have an important effect on water flow through field soils when conditions are such as to maintain a supply of water to the macropores. They suggest this occurs when either the micropore system approaches saturation or when the vertical flow velocities are sufficient to exceed the infiltration capacity of the micropores, either at the surface due to rainfall, or at a permeability break within the soil. The Br pulse was applied when the soil would have been at, or close to, field capacity. Given its non-reactive nature, there was potential for movement of Br in bypass flow to below the depth of cups, before diffusion into the soil matrix had occurred.

One of the deeper (55 cm) cups showed peak Br concentrations only ten days after tracer application. However, the largest daily rainfall between Br application and this peak was 1.7 mm. Such a small event is unlikely to have exceeded the infiltration capacity of the soil surface. Twice during the first two weeks after Br application

porous cups were tampered with, and contamination of the inside of the cup with small volumes of high concentration Br, still on the surface, may explain the anomalously high concentrations in this cup.

Between tracer application and the peak in Br concentrations at 40 cm, there were several days on which precipitation may have caused bypass flow. On 22 March there was 13.2 mm of precipitation, some of which fell as snow. A quick thaw could potentially have exceeded the infiltration capacity of the soil. Similarly, on 17, 18 and 25 April, respectively, 13.6, 15.6 and 16.3 mm of rain fell. The smooth nature of the concentration curves suggests that, if macropore flow did occur, it was not being sampled by the porous cups.

Tyler and Thomas (1981) applied a chloride tracer to undisturbed soil columns and detected maximum concentrations after only 12% of the pore volume equivalent had passed through the soil, indicating two-phase flow. However, Tyler and Thomas (1981) applied the tracer to the surface at a rate just below the soil infiltration capacity.

Another flow route that may not be efficiently sampled by porous cups is lateral flow. Due to the lower hydraulic conductivity of the B horizon, a perched water table frequently develops along the boundary of the A and B horizons, leading to lateral flow.

Porous cups spanned the A/B horizon boundary (section 3.1.1.2) and therefore had the potential to sample at least part of any lateral flow. However, porous cup samples are only representative of the soil solution surrounding the cup at that time (Rhoades and Oster, 1986), and this may change considerably during storm events (Goss *et al.*, 1988). The sampling methodology adopted (section 3.7.3), which was different from the norm (section 2.2.5.2), attempted to maximise the probability of detecting lateral flow. If lateral flow occurred following the rainfall in April, considerable quantities of Br leached from the topsoil may have moved laterally without reaching the depth from which cups were sampling. The subsequent detection of Br may therefore have represented the tail of the Br pulse at much lower concentrations. Barraclough *et al.* (1992) reported that porous cups were not suitable at one site since substantial lateral flow made calculation of the vertical water flux impossible.

Whilst results from the tracer experiment may appear to contradict the overestimation of NO_3^--N concentrations by cups, it is important to consider the source of the solute being studied. Bromide was externally applied to the soil surface (i.e. non-resident) in a concentrated discrete pulse, when the soil was close to field capacity, and therefore the potential for removal without detection by porous cups was high. In contrast, following residue incorporation NO_3^--N evolved within the soil (i.e. resident) over an extended period of time and therefore the NO_3^--N pulse would be more diffuse.

The more erratic NO_3^--N concentrations detected in well samples than in porous cup samples (Figure 4.2.4.1) also suggest that the different sampling methods were sampling different soil water fractions. Well construction aimed to sample the lateral flow of the perched water table at Glencorse, which in itself is an erratic process. During heavy periods of rain, when a perched water table forms, wells sample this fast moving water whereas porous cups only sample some fraction of this water. The proportion of the cup sample consisting of lateral flow depends upon the overlap between the occurrence of lateral flow and the cup sampling period. Drainage water samples provided flow-weighted averages and therefore account for lateral flow when it was the dominant path of drainage water. This explains the general agreement between drain and well samples.

The higher NH_4^+-N concentrations found in well samples than in the porous cups is in agreement with Haines *et al.* (1982) who found 5.1 times more NH_4^+-N in zero tension lysimeters than in tension lysimeters. These authors hypothesised that the difference was due to the different soil water fractions being sampled by the two techniques during rainfall events. An alternative explanation may be that the well samples were contaminated by topsoil knocked into the unlined wells by soil biota (Cuttle, pers. comm).

5.7 Implications of losses of nitrogen following ploughing out of grass-clover swards and possible strategies to decrease these losses

The MAFF Code of Good Agricultural Practice recommends that, if permanent grassland needs reseeding this should be done with as little cultivation as possible, and grass should cover the field by early October (MAFF, 1991). The substantial leaching losses over the first winter from the resown 1992 treatments at Beechgrove (44-50 kg N ha^{-1}) support this recommendation. Calculated NO_3^--N loads in the

Lothian region were in the range of 15-30 kg N ha^{-1} yr^{-1} (DoE, 1994). Clearly, direct reseeding of previously grazed swards produces an above average NO_3^--N input to water courses. However, it must be remembered that reseeding does not take place annually. If it is assumed that reseeding occurs once every 7 years, it would only cause additional leaching losses, averaged for each year, of *ca.* 7 kg N ha^{-1}. Over a 7 year period, the management regime (i.e. grazing or cutting) imposed each and every year becomes a far more important factor than leaching losses incurred in the year of reseeding. Nonetheless, strategies to decrease losses during reseeding should be considered. The two main periods when reseeding occur are spring (March to May) and late summer (July to August), when temperature and moisture are most favourable. The earlier option provides more opportunity for sward establishment, and substantial N uptake, thereby reducing potential leaching losses. This is particularly the case in Scotland where the grass growing season is shorter than in England or Wales.

Where grass-clover swards are incorporated as part of a grass-arable rotation, leaching losses following ploughing out of swards could have a greater impact on average annual leaching losses. Swards are likely to be incorporated more frequently than for direct reseeding, perhaps every 2-3 years. In addition, cereal crops have a shorter growing season than grass (Powlson, 1993). There would therefore be less plant uptake later in the season and any N mineralised after harvest would be readily leached. However, more frequent sward incorporation, in combination with arable cropping, would reduce the accumulation of SOM. This would reduce the amount of N released following ploughing and thereby decrease potential losses.

Given the findings at Beechgrove, sward management prior to ploughing should also be considered. Recent work has reported on the potential reduction in leaching losses that can be made by removing cattle in late summer/autumn to decrease leaching losses from grazed swards (Lord, 1992; Cuttle and Bourne, 1993), in agreement with the MAFF Code of Good Agricultural Practice (MAFF, 1991). Such practices may also be assessed for their potential to manipulate mineral N release patterns following ploughing out of grazed swards. Given the positive effect of excreta on sward residue N inputs, directly and indirectly (section 5.2), it seems likely that continuous grazing until immediately prior to ploughing will produce a rapid release of N, as was observed in 1992. Therefore, by removing grazing animals, and avoiding fertiliser application for a period prior to ploughing, farmers could potentially decrease the N content of sward herbage and remove the supply of N from recently deposited

excreta. This could slow the release of N and match supply more appropriately with the demands of the resown sward, reducing NO_3^--N leaching.

However, the long term build up of SOM quantity and quality is also important (section 5.3). The N released from this source is much harder to control using short term measures. At Beechgrove, the removal of grazing the previous summer resulted in a substantial reduction in the quantity and rate of N release following ploughing in 1993 (section 5.1).

Assuming that excretal returns enhance soil mineral N concentrations for *ca.* 70 days, grazing should therefore cease *ca.* 70 days before the end of the growing season. This would decrease leaching losses over the winter prior to ploughing and, begin the depletion of SOM quantity and quality. The following spring, swards should be subject to a cutting regime, relying on N release from SOM to maintain reasonable productivity prior to ploughing. In this way, the quality of SOM can be manipulated to decrease the amount and rate of N release following ploughing.

Garrett *et al.* (1992) calculated an annual denitrification flux from a grass-clover cattle grazed sward of 6-17 kg N_2O-N ha^{-1} yr^{-1} over three years, with *ca.* 70% of this occurring during autumn and winter. On the grass-clover sward at Beechgrove in 1992, the calculated denitrification flux for the period shortly after ploughing was at the upper end of this range (Table 4.1.4). Even without the reintroduction of grazing animals on the resown sward, reseeding clearly causes a large increase in the annual flux. The management strategies described above for reducing leaching losses could also potentially decrease the losses of gaseous N immediately following residue incorporation. Spring incorporation of residues would be expected to occur at lower soil temperatures and, the depletion of SOM quality would decrease the potential for anaerobiosis.

5.8 Summary

5.8.1 Following ploughing, 38-59% of the total mineral N release observed over eighteen months (*ca.* 370 kg N ha^{-1}) occurred in the first two months. In the first two weeks after rotavation, ammonification was faster than nitrification. In the second cropping season after incorporation, contributions from residues were 19-25 kg N ha^{-1} greater than the release from SOM on the undisturbed swards.

5.8.2 Nitrogen release following ploughing out at Beechgrove was much greater in 1992 than in 1993. This is attributable principally to the cessation of sheep grazing following ploughing out in 1992. This has the effect of:
a) reducing the recycling of N through the animal;
b) reducing the "degradability" of SOM incorporated in the second year.
This was confirmed in an incubation experiment, in which net mineralisation following incorporation was 53% lower from a cut sward than a grazed sward.

5.8.3 Sward clover contents on the Beechgrove trial were 1.6-12.0% of total DM. Differences in N release from the grass and grass-clover sward residues were small. At Glencorse there were larger differences due to the higher sward clover contents (17.7% of total DM) in the grass-clover swards and the low N% of the grass residues (0.8-1.0% N) in the set aside grass swards.

5.8.4 In 1992 both total (1.5-3.7 kg N ha^{-1}) and peak N_2O emissions (485 g N_2O-N ha^{-1} day^{-1}) following sward incorporation were higher than in 1993. This was due to the different weather conditions, the absence of sheep grazing prior to ploughing in 1993 and also changes in plant residue inputs. The importance of grazing was supported by the results of an incubation experiment in which N_2O losses from grazed sward residues, 0.2 kg N ha^{-1}, were twice as high as those from cut sward residues.

5.8.5 Nitrate-N leaching losses, estimated from Beechgrove porous cup data and drain flow data from Glencorse, were 106-250 kg N ha^{-1} from fallow treatments over the first winter following ploughing. These were significantly higher than losses from where grass was resown (44-50 kg N ha^{-1}). There was no significant effect of sward type in the first drainage season following ploughing. Estimated leaching loads for the fallow treatments over the second winter, 200-300 kg N ha^{-1}, were unrealistically high. Evaluation of suction cups at Glencorse in 1993-4 suggests that this method may not produce reliable quantitative estimates of leaching loads.

6 CONCLUSIONS

The ploughing out of grass-clover and fertilised grass swards, previously subject to grazing management, releases a substantial amount of nitrogen in the first eighteen months (*ca.* 370 kg N ha^{-1}). The release of N is greatest during the first two months following ploughing and continues in the second cropping season after incorporation, when contributions from residues were 19-25 kg N ha^{-1} greater than the release from SOM on the undisturbed swards. However, there was evidence to suggest that N supply was limiting productivity of reseeded grass towards the end of the second growing season. A substantial part of the N released following incorporation is from the SOM, though in these trials no direct information regarding the N supply from specific plant fractions or organic matter pools was determined.

The ploughing out of swards subject to an unfertilised cutting regime substantially reduces the release of N following ploughing. This is attributed to the removal of the recycling of N through the animal resulting in a reduction of the "degradability" of SOM incorporated and, a reduction in plant residue N supply. The former is considered to be more important. These findings suggest that there is the potential to manipulate N release patterns following ploughing by altering sward management prior to ploughing.

Where sward clover contents were below *ca.* 10% of total DM, differences in N release from the incorporated grass and grass-clover swards were small. However, where sward clover contents were approaching 20% of total DM, and grass residues were of lower N content, clover residues had a more significant impact on mineral N release. The effect of clover residues on N_2O emissions was also dependent on the sward clover content. However, the potential creation of 'hotspots' of N_2O emission means that the potential influence of clover residues on gaseous emissions is greater than their potential influence on mineral N release patterns.

The ploughing out of grassland produces a considerable short term input of N_2O to the atmosphere due to the supply of readily available carbon. Nitrous oxide flux patterns suggest that rotavation, rather than ploughing, initiates the increase in emissions. This is thought to be due to the more intimate mixing of plant residues into the soil by rotavation, thereby increasing the potential for anaerobic 'hotspots'. The removal of grazing sheep for a period prior to ploughing has the potential to reduce such emissions considerably. However, our ability to reduce such detrimental

environmental effects is limited by the external influence of the weather at critical times following incorporation.

The potential for NO_3^--N leaching losses over the first winter following the ploughing out of grazed swards were substantial (106-250 kg N ha^{-1}) if no subsequent crop was sown. Whilst the reseeding of a grass sward significantly reduced leaching, losses were still much greater (44-50 kg N ha^{-1}) than from the undisturbed cut swards, which were negligible. The lowest leaching losses following incorporation occurred when unfertilised, cut swards were ploughed out earlier in the year and resown to either spring barley or grass (<1-13 kg N ha^{-1}).

Evaluation of estimated leaching loads over the second winter, 1993-4, and specific tests comparing porous cups with three other estimates of soil water NO_3^--N concentrations at Glencorse in 1993-4, cast serious doubts as to the quantitative accuracy of porous cups on medium textured soils. Quantitative data rather than merely comparative measures of NO_3^--N loss are necessary, in order to assess whether losses exceed a particular environmental target (Lord and Shepherd, 1993). More comprehensive testing of porous cups on such soils is required in order to clarify exactly what fraction of the soil solution cups actually sample. Results from the field trials suggest that cups were sampling relatively immobile water and, depending on the source of the solute, this led to over- and under-estimates of leaching loads.

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APPENDICES

APPENDIX 1 Bromide tracer experiment

1.1 Assumptions in calculating estimated vegetation uptake of bromide in the bromide tracer experiment

Owens *et al.* (1985) found that the concentration of Br in grass, five weeks after application of 16.8 g Br m^{-2}, was 1.65%. Given that 5 g Br m^{-2} was applied at Beechgrove, and assuming similar soil bulk densities in the two studies, the soil Br concentration at Beechgrove would be 30% (i.e. 16.8 / 5) of that in the study of Owens *et al.* (1985). Using the linear relationship between soil Br concentration and plant Br concentration (Kempton and Maw, 1973), the Br concentration in grass at Beechgrove would be estimated to be 0.5% (i.e. 1.65% x 0.3). Given that Owens *et al.* (1985) observed a grass Br concentration of 0.6% in hay one year after Br application, 0.5% seems an underestimate for the concentration of Br in grass at Beechgrove. This is supported by Brown *et al.* (1958) who found that Br uptake increased with increasing clay content in the soil, since the soil at Beechgrove was a clay loam whilst the study of Owens *et al.* (1985) was on a silty loam. Consequently, it was assumed that the initial concentration of Br in grass at Beechgrove was 0.7%, declining through the season as soil Br concentration decreased due to plant uptake and leaching.

1.2 Equation used to estimate the depth of Br tracer pulse displacement

Equation A1.2

$$Zp = \frac{Q}{\theta_v}$$

where

Zp = the depth of penetration of the displacing solution (mm)

Q = rainfall (mm) = 146.2 mm between tracer application and the peak in Br concentrations in porous cups

θ_v = volumetric moisture content at field capacity (cm^3 H_2O cm^3 $soil^{-1}$)

= 0.41 and 0.35 cm^3 H_2O cm^3 $soil^{-1}$ for the 0-20 and 20+ cm layers, respectively. (Assuming dry bulk densities of 1.17 and 1.4 g cm^{-3}, and gravimetric moisture contents at field capacity of 0.35 and 0.25 g H_2O g $soil^{-1}$, for the 0-20 and 20+ cm layers, respectively.)

APPENDIX 2 Assumptions taken in calculating vegetation uptake of nitrogen on the Beechgrove field trial, 1992-3

2.1 The estimate of DM uptake due to grazing, prior to the blocks being fenced off on 22 July 1992, was based on the number of days on which sheep were observed in the surrounding paddocks and, measured DM production rates during subsequent sampling. A maximum of 27 days grazing took place and, it was assumed that 300 and 800 kg DM ha^{-1} was removed by grazing between ploughing and fencing on the grass and grass-clover swards, respectively.

2.2 The DM production of regrowth on the fallow treatments was assumed to have occurred in a linear fashion between 7 August and sampling on 15 October 1992. This starting point was chosen because the second rotavation did not take place until 20 July, and only on 11 August was regrowth noticeable.

2.3 Prior to the ploughing of the resown 1993 treatments, vegetation data for the undisturbed and resown 1993 treatments were used interchangeably in their respective sward blocks.

2.4 Where appropriate data was unavailable, vegetation data were interpolated linearly between sampling dates. The exceptions to this were:

a) Data from 7 August 1992 and 22 October 1993 was assumed to apply to the pre-fencing and 17 December 1993 sampling dates, respectively.

b) Sward clover contents for 9 August, 22 October and 17 December 1993 were assumed to be equal to the average of sward clover contents from the three previous sampling dates (30 March, 4 May and 6 June).

2.5 Estimation of regrowth between 22 October and 17 December 1993.

The daily DM production rate was calculated for the previous sampling period (9/17 August-22 October). The DM production rate (under optimum conditions) in October is roughly half that in September. Therefore the DM production rate between 22 October and 17 December was likely to be less than this, say 25%.
In order to verify this rough estimate, a more sophisticated estimate of DM production was calculated based on studies of Keatinge *et al.* (1979) during winter in

Northern Ireland. Keatinge *et al.* (1979) found the linear relationship between temperature and leaf extension of perennial ryegrass shown in equation A2.5.1.

This calculation suggested that the DM production rate between 22 October and 17 December was about 15% of the DM production rate between 9/17 August-22 October. Thus regrowth since 22 October was calculated using equation A2.5.2.

Equation A2.5.1

$$LER = 0.35 + 0.51\,T$$

where:
LER = leaf extension rate (mm day^{-1})
T = mean daily air temperature

Equation A2.5.2

$$\text{Regrowth} = DGR_{t-1} \times K \times N$$

where:
DGR_{t-1} = daily DM production rate for the previous sampling period (9/17 August-22 October).
K = calculated DM production rate conversion factor = 0.15
N = number of days between 22 October and 17 December = 56

2.6 Estimation of DM uptake by resown grass below the cutting height.

Total DM uptake below the cutting height was calculated using equation A2.6 below and the coefficients in Table A2.6. These coefficients were based on estimates of grass development from field notes on shoot height, appearance, development stage, photographic evidence and supplementary data from studies of ryegrass establishment (Jones and Lazenby, 1988).

Equation A2.6

UNCUT DM between t and t-1 = $((TT_{717} - R) \times 0.65^{a} \times C) + ((TT_{717} - R) \times D)$

where:

t = sampling date

t-1 = previous sampling date

UNCUT DM = Total DM uptake below cutting height in the establishment year

TT_{717} = December 17 turf tops

R = regrowth between 22 October and 17 December (as calculated in equation A2.5.2)

D = Fraction of leaves which were dead

C = Ground cover as a fraction of ground cover on 17 December 1993

[a] Robson (1973) calculated that dead leaves account for about 35% of total ryegrass biomass after 12 weeks when grown under optimal conditions.

Table A2.6 Coefficients used for each vegetation sampling date in equation A2.6.

	Resown 1992 treatments		Resown 1993 treatments	
Sampling date	D	C	D	C
16 September 1992	0	0.6	-	-
15 October 1992	0.2	1.0	-	-
30 March 1993	0.35	1.0	-	-
9/17 August 1993	-	-	0	0.6
22 October 1993	-	-	0.25	1.0
17 December 1993	-	-	0.35	1.0

2.7 Estimation of root DM uptake in ploughed treatments

Root DM was not sampled on any ploughed plots until 17 December 1993. Assumptions regarding root DM development in the first year following sowing were needed in order to provide an estimate of total vegetation DM uptake on the ploughed treatments.

Firstly, calculation of new root development (defined as those roots belonging to the resown ryegrass as opposed to old, undecomposed MOM remaining from ploughed

out sward residues) was necessary. Thus MOM data from control plots (ploughed but not resown) were required to estimate the remaining undecomposed MOM. For treatments ploughed out in 1992, fallow treatments in the respective blocks were used as controls. For treatments ploughed out in 1993, no direct controls were available. Instead MOM data from an area of grass-clover sward sprayed with Glyophosate on 23 December 1992 was used. Samples from the control areas were taken on 28 April 1994. It was assumed no further decomposition had occurred between 17 December 1993 and 28 April 1994 given that soil temperatures rarely rose above 5°C.

Clearly, the control area for treatments ploughed out in 1993 included potential decomposition of residues between 23 December 1992 and 11 May 1993 which residues in the resown 1993 treatments would not have been subjected to. To account for this, meteorological data was used to calculate 'potential decomposition time' (Equation A2.7) for the periods between 23 December 1992 and 11 May 1993 and, 23 December 1992 and 17 December 1993. It is assumed no decomposition occurs below 0°C and decomposition rate has a Q_{10} of 3 between 5 and 15°C.

Equation A2.7:

$$\text{Potential decomposition time} = \frac{T}{5}$$

where:
T = mean of soil temperature at 10 cm and 30 cm

Cumulative potential decomposition time for the period between 23 December 1992 and 11 May 1993 was 21% greater than between 23 December 1992 and 17 December 1993. Actual decomposition, calculated by subtracting control MOM (sprayed area) from the total ploughed in residue DM (Table 4.1.2.2), was therefore reduced by 21% to provide an estimate of 'true' control MOM.

Having estimated total root DM uptake using MOM analysis, a second estimate of root DM uptake, based on the work of Troughton (1956), was calculated. Troughton (1956) described the relationship between shoot and root DM production for 12 weeks following the sowing of grass. Thus, calculated figures for shoot DM production could be used to calculate root DM production. It was assumed that on

the date of the first cut, approximately two months after ploughing, root DM production was half that of shoot production. Having calculated root mass at this first sampling date, Troughton's pattern of root mass increase was applied to estimate further root DM production. In 1992 it was assumed that root mass doubled between the first and second vegetation cuts and, by 30 March 1993, was 3.5 times the level at the first cut. In 1993 it was assumed that root mass trebled between the first and second vegetation cuts, and by 17 December 1993 was 3.5 times the level at the first cut. Data calculated using this method compared favourably with the MOM analysis.

MOM data showed large variability and root mass estimates using this estimation technique showed no clear relationship with shoot DM. The Troughton estimation technique was dependent on shoot DM production data which was more reliable and less erratic than MOM data. It also provided an easier technique for temporal distribution of root growth. Therefore, the Troughton estimation technique was used.

On fallow plots, the natural regrowth which occurred was also assumed to have root mass equivalent to half of its shoot mass.

In order to calculate root N uptake one root N content, 1.36%, was assumed to apply to all roots at all times unless otherwise stated. This figure was based on data for roots on 31 March and 17 December 1993.

APPENDIX 3 The effect of suction on the ammonium-N and nitrate-N concentration in porous cup water samples

Table A3 Ammonium-N and nitrate-N concentrations in porous cup samples using 0.2 and 0.7 bar on the Beechgrove trial, 1992.

Treatment	Cup no.	Sampling dates[a]	Nitrate-N ($\mu g\ N\ ml^{-1}$)[b]		Ammonium-N ($\mu g\ N\ ml^{-1}$)[b]	
Suction applied (bar)			0.2	0.7	0.2	0.7
PGC 92	22	12 & 14 August	**22.32**	18.76	0.63	**0.83**
	24	12 & 14 August	**59.52**	57.28	0.16	**0.47**
	28	12 & 14 August	**44.28**	39.91	0.34	**0.47**
	24	16-18 September	49.28	**54.25**	**0.59**	0.24
	28	16-18 September	19.35	**21.8**	-	-
	24	23-25 September	**27.8**	11.4	-	-
	28	29 Sep-1 Oct	**6.05**	4.9	-	-
	24	7-8 October	3.15	**3.4**	-	-
	28	7-8 October	**2.05**	0.79	-	-
PGCF	25	12 & 14 August	**92.64**	71.58	0.5	**0.63**
	26	12 & 14 August	**114.24**	82.51	0.23	**0.56**
	27	12 & 14 August	**68.16**	64.63	0.11	**0.27**
	25	16-18 September	188	**198.5**	**0.33**	0.25
	26	16-18 September	**161.28**	127.6	-	-
	26	23-25 September	**144.0**	143.2	-	-
	27	29 Sep-1 Oct	**131.2**	115.2	-	-
	25	7-8 October	104.8	104.8	-	-
	26	7-8 October	**109.6**	101.2	-	-
CGC	23	12 & 14 August	1.79	**4.61**	0.36	0.36
	20	16-18 September	.11	**0.89**	0.62	**0.75**
	21	16-18 September	**6.45**	3.62	-	-
	23	23-25 September	**0.32**	0.02	-	-
	21	29 Sep-1 Oct	0.0	**0.39**	-	-
PGF	29	16-18 September	**147.6**	144.8	**1.18**	0.75
	3	16-18 September	53.44	**55.35**	-	-
	29	23-25 September	169.6	**182.9**	**0.67**	0.51
	3	29 Sep-1 Oct	73.2	**80.0**	-	-
	10	7-8 October	**100.4**	92	-	-
	29	7-8 October	**198**	183	0.42	**0.43**
	3	7-8 October	84	**92.4**	-	-

[a] Sampling took place over several days and therefore the range is given.
[b] The suction yielding the higher concentration for each sampling occasion is shown in bold type.

APPENDIX 4

Inter- and intra-plot variability of soil water nitrate-nitrogen concentrations in porous cup water samples

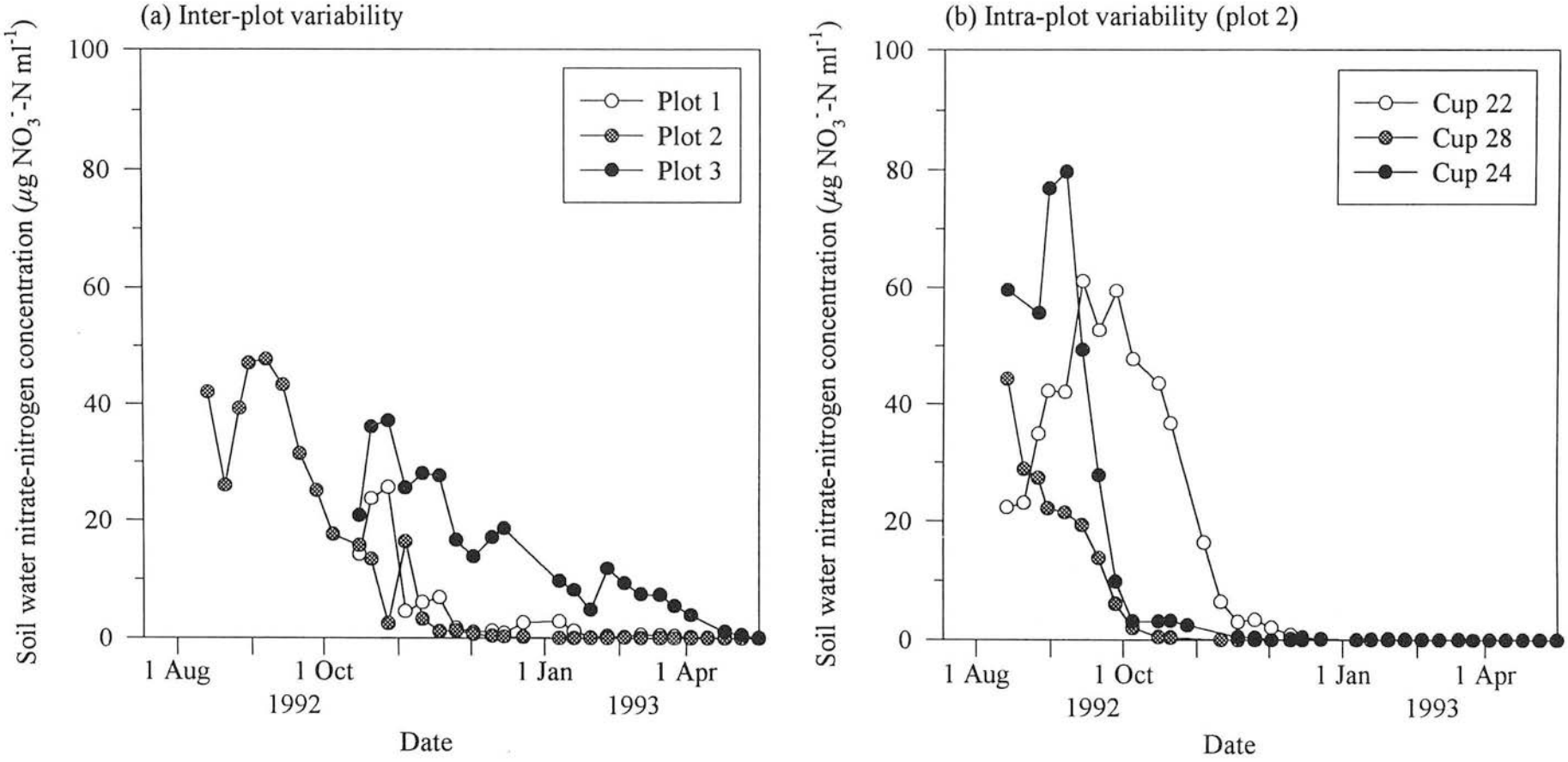

Figure A4.1 Soil water nitrate-nitrogen concentrations in porous cup water samples in the ploughed out grass-clover resown 1992 treatment (a) inter-plot variability (b) intra-plot variability.

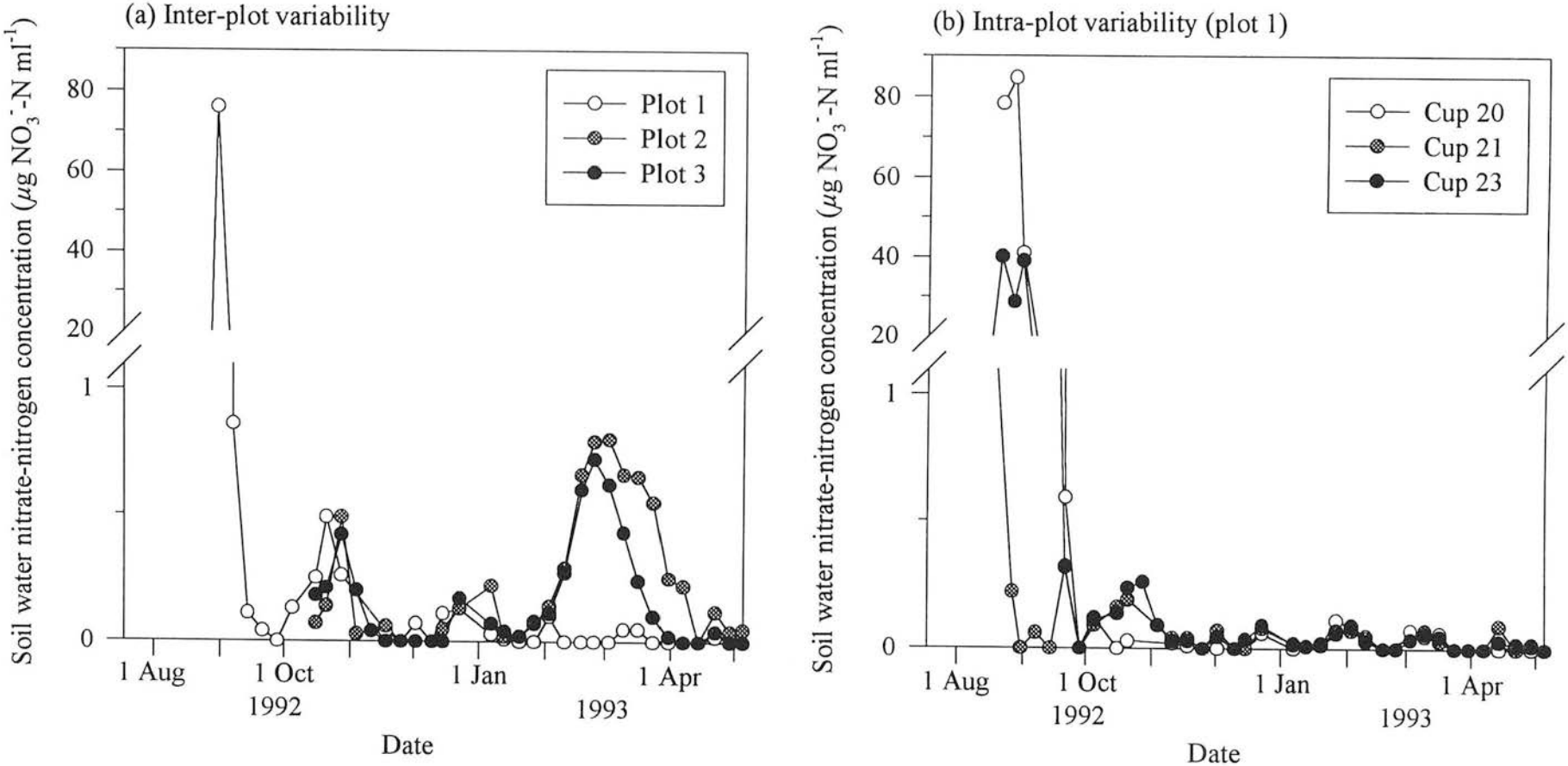

Figure A4.2 Soil water nitrate-nitrogen concentrations in porous cup water samples in the continued grass-clover treatment (a) inter-plot variability (b) intra-plot variability.

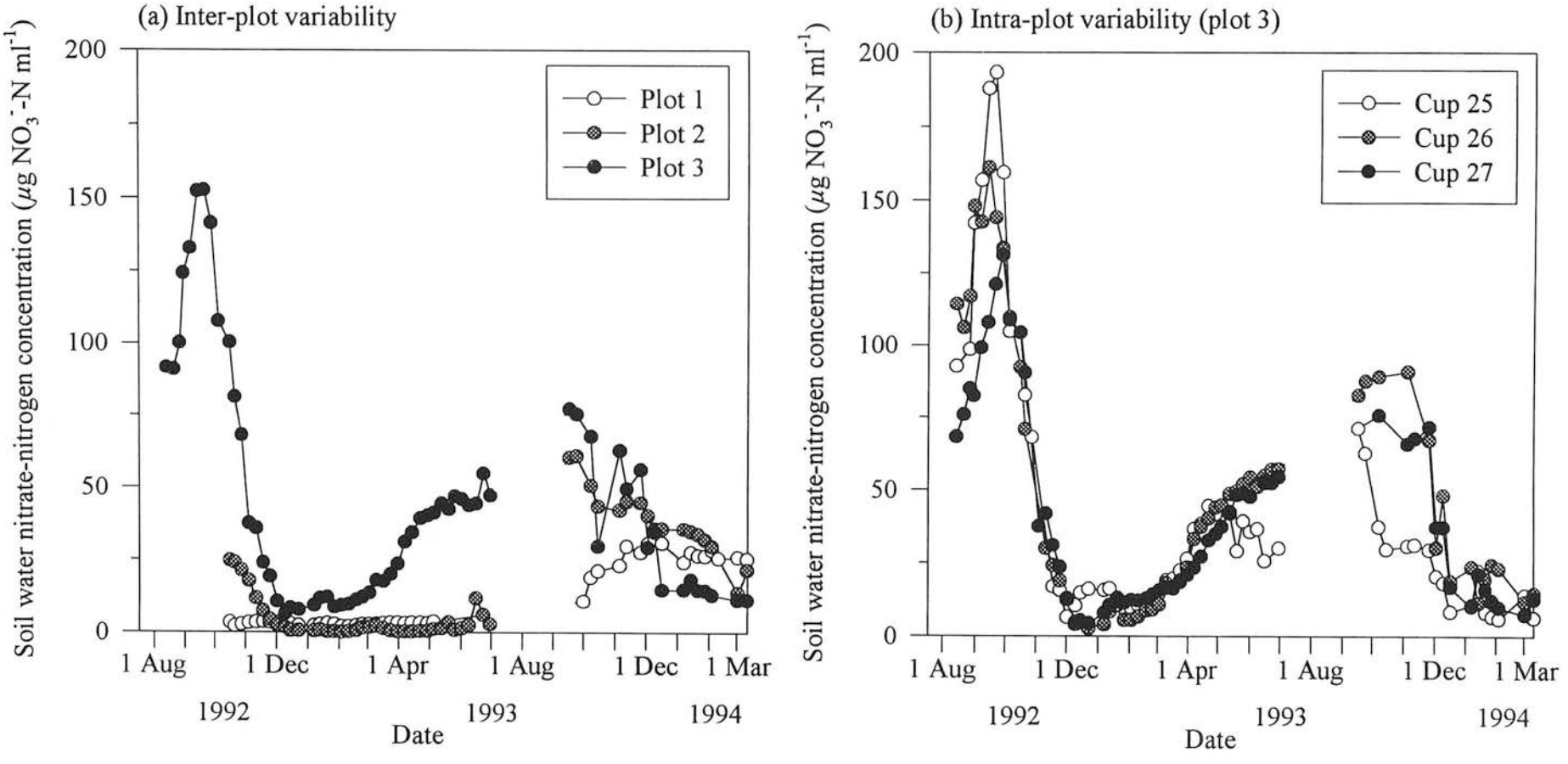

Figure A4.3 Soil water nitrate-nitrogen concentrations in porous cup water samples in the ploughed out grass-clover fallow treatment (a) inter-plot variability (b) intra-plot variablity.

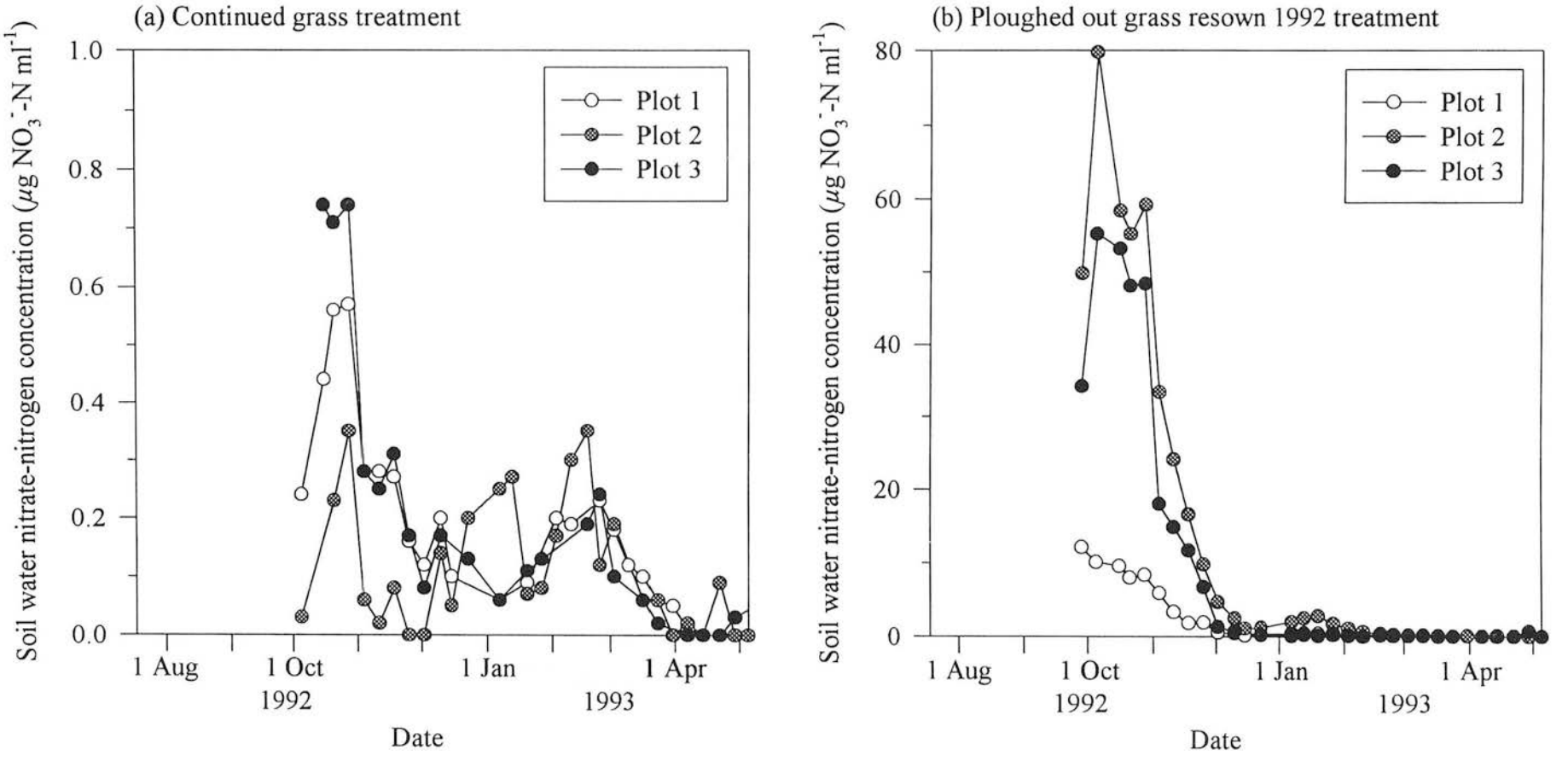

Figure A4.4 Inter-plot variability of soil water nitrate-nitrogen concentrations in porous cup water samples (a) Continued grass treatment (b) Ploughed out grass resown 1992 treatment.

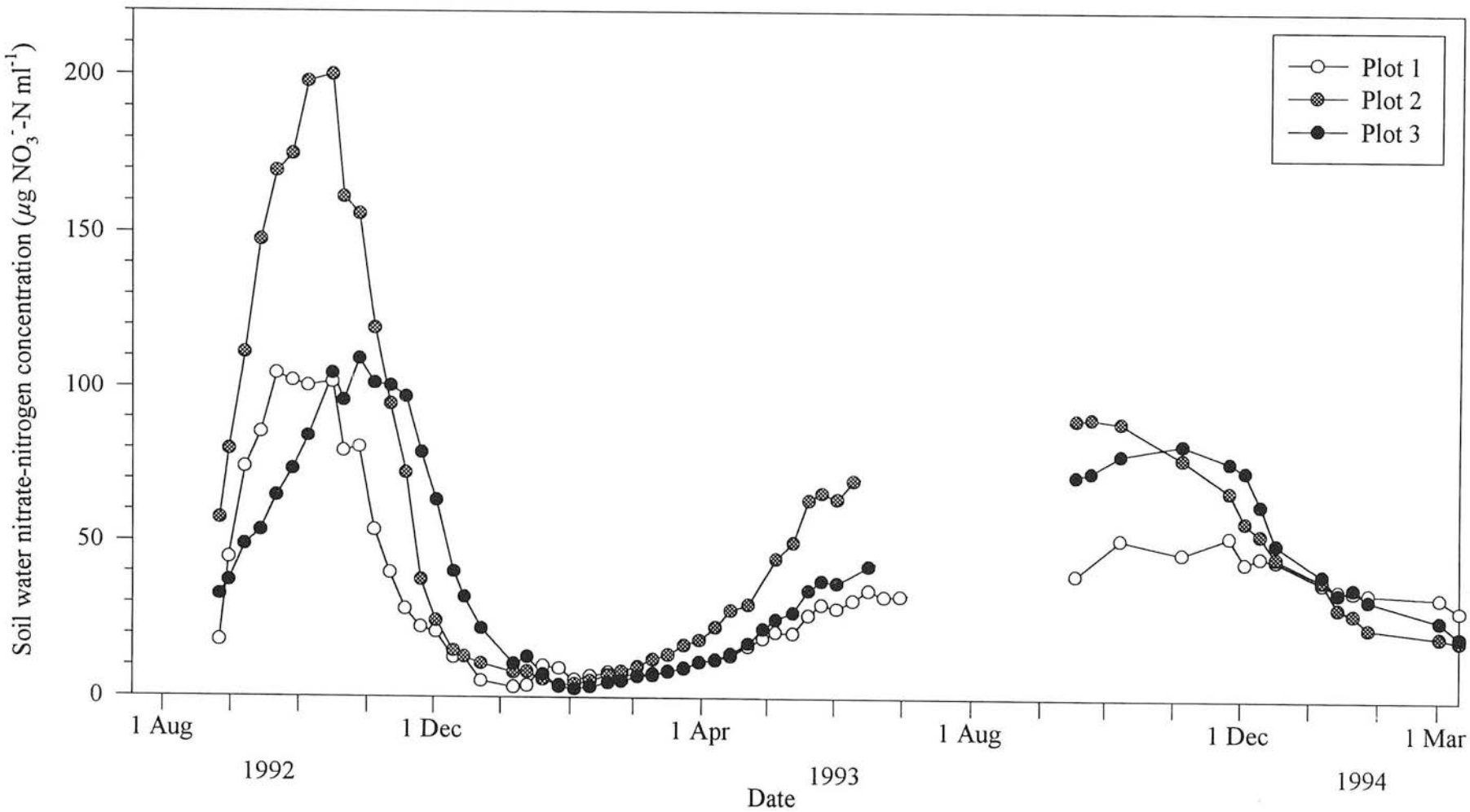

Figure A4.5 Inter-plot variability of soil water nitrate-nitrogen concentrations in porous cup water samples in the ploughed grass fallow treatment

APPENDIX 5 Equations for calculating N_2O yields

Equation A5.1

$$N_2O \text{ yield in } \% = \frac{N_2O\text{-}N_{(f)} - N_2O\text{-}N_{(c)}}{\text{N application}} \times 100$$

where:

$N_2O - N_{(f)}$ = flux from fertilised plots

$N_2O - N_{(c)}$ = flux from unfertilised control plots over a variable, but extended period of time.

Equation A5.2

$$N_2O \text{ yield in } \% = \frac{N_2O\text{-}N_{(f)}}{\text{N application}} \times 100$$

where:

$N_2O - N_{(f)}$ = flux from fertilised plots.

APPENDIX 6

Assessment of the problems associated with measuring gross mineralization rates in soil cores

Assessment of the problems associated with measuring gross mineralization rates in soil cores

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Abstract

Intact soil cores from three sites were injected with a small amount of 99 atom% enriched ammonium sulphate and incubated at a range of temperatures or moisture potentials for between 1 and 40 days in the laboratory. The experiments were aimed at assessing the problems associated with measuring field rates of gross mineralization in intact cores, following work done by Davidson *et al.* (1991).

Very large drops in the $^{15}NH_4^+$ were seen in the first 24 hours after injection, which were not characteristic of the whole incubation period. This rapid fall may be caused by slow equilibriation of NH_4^+ between the fixed and exchangeable pools. The added $^{15}NH_4^+$ and pre-existing NH_4^+ pools may also be subject to different consumption rates during this period due to the incomplete equilibrium and spatial separation of the two pools. Recycling of ^{15}N was indicated in our experiments by an increase in the amount of $^{15}NH_4^+$. This was seen after seven days at 14 °C, and took longer to occur at the lower temperatures.

Use of pool dilution methods must be accompanied by a check that the underlying assumptions are valid and experimental procedures should be chosen to minimize error. Spatial variability of mineral nitrogen in the field can lead to inaccuracies in calculated rates. Where the method is used with care, however, it will lead to a greater understanding of the processes of the soil nitrogen cycle.

INTRODUCTION

The soil nitrogen cycle involves a large number of identifiable pools of nitrogen linked by complex, often simultaneous and opposing, processes (Jansson, 1958; Paul and Juma, 1981). Simple observations mapping the sizes of the nitrogen pools involved over time are therefore not adequate to describe the full dynamics of the system (Jansson and Persson, 1982). Use of ^{15}N has provided a powerful tool, where direct tracer methods determine the location of ^{15}N after a period of exposure and pool dilution methods estimate flow rates

In: J.J. Neeteson and J. Hassink (eds.), 1994. Nitrogen mineralization in agricultural soils; proc. symp. held at the Institute for Soil Fertility Research, Haren, NL, 19-20 April 1993. AB-DLO Thema's, AB-DLO, Haren, pp. 95-110.

through a given nitrogen pool (Nason and Myrold, 1991). Where labelled NH_4^+ is added to the NH_4^+ pool in the soil and comes rapidly into equilibrium with it, the decline in the ^{15}N enrichment of the pool, as NH_4^+ at natural abundance is introduced by mineralization of soil organic nitrogen, can be used to calculate the gross mineralization rate. In the same way addition of labelled NO_3^- can allow measurement of gross nitrification rates.

As in all isotopic work, it is necessary to assume that ^{14}N and ^{15}N are not discriminated by processes occurring in soils and that the added ^{15}N equilibriates rapidly with the pool to which it has been added, creating a single homogeneous pool. Whilst some microbial transformation processes do discriminate between ^{14}N and ^{15}N (Cheng *et al.*, 1964; Heaton, 1986), the assumption probably holds for incubations of enriched samples over a short time period. The heterogeneity of ^{15}N in the organic nitrogen pool has not been studied and it is assumed that the ^{15}N abundance is naturally at background levels (Wessel and Tietema, 1992). There are some indications from plant uptake studies (McTaggart, 1992) that the ^{15}N natural abundance of the active pool of soil organic nitrogen is higher (0.370-0.377 atom%) than the value of 0.3663 atom%, and where possible the ^{15}N natural abundance should be measured at the study site.

When experiments are carried out in the laboratory with sieved and well mixed soils or litters (e.g. Bjarnason, 1988; Wessel and Tietema, 1992) it is relatively easy to achieve uniform addition of ^{15}N. However, in the field it is almost impossible to achieve such uniform applications (Barraclough, 1991; Davidson *et al.*, 1991). The heterogeneity of soils in the field, particularly the spatial variability of mineral nitrogen, is a serious problem for the development of *in situ* applications of isotope dilution, since rate estimates are calculated using differences, which amplify errors (Myrold and Tiedje, 1986).

To simplify calculations, mineralization and immobilization rates are usually assumed to be constant (Kirkham and Bartholomew, 1954; Blackburn, 1979), or to vary according to some known relationship (Nason and Myrold, 1991), between measurements of pool size and enrichment. The change in the ^{15}N enrichment of the NH_4^+ pool as mineralization and consumption processes proceed is complex and simple averages of the ^{15}N enrichment between measurements (Shen *et al.*, 1984; Guiraud *et al.*, 1989) can only give approximations of gross rates. The enrichment of ^{15}N only declines linearly for very short periods of time even where mineralization and immobilization are proceeding at constant rate (Bjarnason, 1988).

Table 1. List of symbols used in the equations.

AT_1	Total size of NH_4^+ pool, μg N g^{-1}, at time 1.
AT_2	Total size of NH_4^+ pool, μg N g^{-1}, at time 2.
AL_1	Size of labelled NH_4^+ pool, μg N g^{-1}, at time 1.
AL_2	Size of labelled NH_4^+ pool, μg N g^{-1}, at time 2.
t	Time between measurements, days.
@	Natural ^{15}N enrichment of mineralising NH_4^+.
m	Rate of mineralization / production of NH_4^+, μg N g^{-1} day^{-1}.
c	Rate of NH_4^+ consumption, μg N g^{-1} day^{-1}.

A formal mathematical treatment to allow the calculation of gross mineralization and consumption rates has existed since 1954 (Kirkham and Bartholomew, 1954), where the changes in pool sizes are described by differential equations, and solved analytically. Remineralization of immobilized mineral nitrogen is disregarded and the change in

amount of ^{15}N in the NH_4^+ pool is derived only from the consumption process. Symbols are defined in Table 1.

$$\frac{d\,AL}{d\,t} = -c\,\frac{d\,AL}{d\,AT}$$

$$\text{Then} \qquad m = \frac{(AT_2 - AT_1)}{t}\,\frac{\log (AL_1AT_2/AL_2AT_1)}{\log (AT_2/AT_1)}$$

This framework is only valid where ^{15}N addition to the soil is high and where the ^{15}N enrichment of the NH_4^+ pool does not approach background by the end of the incubation period. However, it is still widely used for the calculation of gross mineralization rates (e.g. Davidson *et al.*, 1991; Ambus *et al.*, 1992). The model was extended to allow for nitrogen mineralizing at natural or any fixed ^{15}N abundance from the organic nitrogen pool for anoxic sediments (Blackburn, 1979) and for aerobic soils (Nishio *et al.*, 1985), deriving the decline in ^{15}N enrichment from the consumption and mineralization processes, but still not accounting for remineralization. Symbols are defined in Table 1.

$$\frac{d\,AL}{d\,t} = @\,m - c\,\frac{d\,AL}{d\,AT}$$

$$\text{Then} \qquad m = \frac{(AT2 - AT1)}{t}\,\frac{\log ((AL2/AT2)-@\,/(AL1/AT1)-@)}{\log (\,AT2\,/\,AT1)}$$

These equations can be applied to $^{15}NO_3^-$ as well as $^{15}NH_4^+$ additions (Schimel *et al.*, 1989), allowing calculation of both gross mineralization and nitrification rates. A calculation method allowing calculation of gross nitrification rates where only $^{15}NH_4^+$ is added has also been developed (Wessel and Tietema, 1992).

Kirkham and Bartholomew (1955) developed a second mathematical framework allowing for nitrogen mineralizing at natural abundance and possible remineralization of added labelled nitrogen, by estimation of the interacting organic nitrogen pool. However, this model was developed for a simple system of two pools with mass conservation assumed and it cannot be corrected for losses to the NH_4^+ pool other than by immobilization to organic nitrogen.

Numerical solutions of the differential equations have also been developed, where numerical simulation by use of non-linear curve fitting, uses the measured ^{15}N abundances in the mineral N pool to fit the gross transformation rates and the size of the initial organic nitrogen pool involved (Myrold and Tiedje, 1986; Barraclough and Smith, 1987; Bjarnason, 1988). The advantages of numerical solutions are that they can be applied to any set of differential equations and the solution procedure remains the same, irrespective of the chosen set of rates, pools and other conditions (Wessel and Tietema, 1992). However, a high degree of replication is required to fit a solution with any degree of certainty and analytical models offer a quick way to calculate gross rates, so long as their assumptions have not been violated.

A study was carried out on three soil types to investigate the potential problems in applying the method used to measure gross mineralization rates described by Davidson *et al.* (1991). Concentrations of NO_3^- and NH_4^+ in injected and uninjected cores were compared to establish if the addition of $^{15}NH_4^+$ solution provided a stimulus to mineralization.

Incubations were carried out for increasing periods to determine an appropriate length of field incubation under Scottish temperature regimes. Observations were made of reminer-alization on one soil type, whilst the effect of soil moisture potential on mineralization was observed in intact cores of the other soils. Some preliminary work was carried out to establish an appropriate injection methodology with minimum disturbance to the core, and to determine the variability of immediate NH_4^+ fixation within a soil type. A further experiment attempted to assess the replication necessary to allow estimates of the mean mineral nitrogen pool size.

MATERIALS AND METHODS

Core sampling and core preparation

Intact soil cores, 54 mm in diameter and 20 cm deep, were sampled inside PVC sleeves using a specially designed corer at three sites on the Bush Estate, 15 km south of Edinburgh, in winter 1991-1992. Properties of the soils used in the study are presented in Table 2. The cores from the Beechgrove site were roughly crumbled and the sward torn into pieces in an attempt to simulate ploughing. The 'ploughed' soil was then packed back into the PVC sleeve with the sward distributed randomly throughout. The core was pushed out of the liner into a polyester sock to enable good moisture equilibriation during incubation. The cores from the Glencorse and No. 3 sites were left intact for the incubations. A further 100 samples were taken from the Beechgrove site, three days after ploughing in the spring, using an Dutch auger, to allow assessment of the spatial vari-ability of mineral nitrogen at the site at a time when mineralization measurements in the field were likely to be carried out.

Table 2. Properties of the soils used (0-20cm).

Site	Glencorse	No. 3 field	Beechgrove
Topsoil texture	clay loam	sandy loam	sandy loam
O.M. %	5.0	4.0	4.8
pH	6.4	6.3	6.0
Crop	cereals	cereals	pasture
Soil series	Winton	Macmerry	Winton

Core Injection

Several preliminary tests were carried out using an iodine-green dye solution to assess the most suitable method of injection and appropriate points of injection into the core. A visual assessment of the distribution of the injected solution was made by slicing the core horizontally at 2 cm intervals and studying the cross-sections. Preliminary extractions were carried out on twenty cores from the Beechgrove site to establish how much $^{15}NH_4^+$ should be added to each core to give an NH_4^+ pool with an enrichment of approximately 25 atom%. Following this work, five ml (measured gravimetrically) of 0.17 g l^{-1} $(^{15}NH_4)_2SO_4$ solution of 99.2 atom% enrichment, containing 190.06 μg ^{15}N, was injected into the core at five points using 1 ml syringes, which penetrated approximately 3 cm into the core. As

the solution was injected the needle was slowly withdrawn to enhance the distribution of solution through the core.

Core incubation

The cores from the Beechgrove site were incubated in sand tanks at a range of constant temperatures (4, 10, 14 °C), chosen to reflect the seasonal range of temperatures in Scotland, for a number of incubation periods (1, 2, 4, 7, 11, 14 and 18 days). The tanks were filled with 32 cm depth of coarse sand, which had been saturated and left to drain to give a water table 1 cm above the base of the tank and 11 cm below the base of the cores. This water level was maintained using a simple gravity fed constant head device, and enabled the cores to be held at constant moisture potential through the incubation period. The cores were randomly divided into three groups and allowed to equilibriate for four days at each temperature before any cores were injected.

Cores of the Beechgrove soil were placed into the sand tanks into auger holes of an appropriate size and the sand was tamped to ensure a good core-sand contact. For each temperature and each incubation period, three cores were injected with $(^{15}NH_4)_2SO_4$ solution and four replicate cores were used for assessment of net mineralization rates. The position of the cores was randomized in the tank, with cores 4 cm apart. Core temperatures were measured on each sampling date, using a temperature probe inserted through the centre of each core, prior to its removal from the tank. Gravimetric moisture contents were calculated for each core after removal, by oven drying at 105 °C for 24 hours.

The cores from the Glencorse and No. 3 sites were incubated at room temperature, approximately 18 °C, at a range of matric suctions (300, 100, 10 and 1 kPa) for three periods of time (7, 18, and 40 days). The 300 kPa and 100 kPa moisture potentials were obtained using a pressure membrane apparatus. The 10 kPa moisture potential was set up using a tension tank and this apparatus was modified to provide the 1 kPa potential. Four replicate cores for each matric tension and time period were injected with $(^{15}NH_4)_2SO_4$ solution, after equilibriation at the appropriate moisture potential. The gravimetric moisture contents of the cores were measured at the end of each time period.

Nitrogen analysis

Available NH_4^+ and NO_3^- were determined by extracting soil using 1 M KCl in a 1:5 soil:solution ratio. Extracts were filtered through Whatman 42 filter paper and NO_3^--N and NH_4^+-N determined by continuous flow analysis (Best, 1976; Crooke and Simpson, 1970). Four cores from each temperature or moisture potential were extracted at the beginning of the incubations. Analysis of injected cores for $^{15}NH_4^+$ enrichment was carried out by steam distillation followed by mass spectrometric determination as described by Hauck (1982). Where NH_4^+-N concentrations in the extracts were < 500 μg, a carrier solution containing 1 mg N as NH_4^+ was added prior to steam distillation to ensure that enough N was present to allow determination of the ^{15}N abundance.

To allow for fixation of the NH_4^+ in the soil, a recovery factor was calculated by extracting the NH_4^+ pool of the soil 15 minutes after injection and determining its ^{15}N enrichment (Davidson *et al.*, 1991). The recovery was calculated as the proportion of the added $^{15}NH_4^+$, 190.06 μg ^{15}N per core, which could be extracted from the soil in the NH_4^+ pool using KCl. This was carried out for the Beechgrove and Glencorse soils. A recovery factor was also calculated for the No. 3 and Glencorse soils where extractions took place

within twenty four hours of injection, and a comparable 24 hour recovery factor could be calculated from the incubations with the Beechgrove soil. No direct measurements of NH_4^+ clay fixation were made, and the precise clay mineralogy was difficult to estimate from the literature since the glacial till from which the soils are derived is very variable.

Mathematical models

The analytical model developed by Blackburn (1979) was used to calculate the gross mineralization and consumption rates for the system, where remineralization could be neglected. As no value for the natural ^{15}N enrichment of the active organic pool was available, a natural abundance of 0.3663 atom% was used. This may have led to slight underestimates of the rates of gross mineralization.

Statistical methods

Differences between treatments were compared using oneway Anova procedures and injected and uninjected cores compared using t-tests. All analyses were carrried out using the MINITAB stastistics package.

RESULTS AND DISCUSSION

Injection method

Preliminary tests revealed that one large injection often led to macropore flow out of the core, whilst several smaller injections of smaller volumes gave a better distribution of solution. It was therefore decided that five injections of 1 ml should be made to minimize leaching straight through macropores and maximise the proportion of the core volume receiving the solution. One injection was made centrally at the top of the core followed by four injections into the sides at approximately 5, 9, 13 and 17 cm from the top of the core. Further tests on four cores using this injection procedure found that 27 % of the cross-sections showed no visible signs of dye and there were clear signs of preferential flow of the solution along macropores.

Recovery

After 15 minutes, only 61.4 % on average of the $^{15}NH_4^+$ injected into the Beechgrove soil was recovered in a KCl extract, i.e. the mean recovery factor was 0.614 (S.E. 0.0366, 12 replicates). In the Glencorse soil, the corresponding recovery factor was 0.42. The processes leading to this disappearance did not seem to be highly spatially variable for any one soil series, but recovery was significantly different between soils ($P < 0.05$). The recovery factor was significantly ($P < 0.001$) affected by soil moisture (Figure 1), with the values decreasing in the drier soils ($r = 0.854$; $df = 22$). The relationship seemed to hold over a relatively wide moisture range, irrespective of soil texture. Injected solution was subject to greater soil aggregate suction in the drier soils, hence in these soils the solution may have penetrated further into the soil aggregates and possibly encountered more fixation/consumption sites in the same time period.

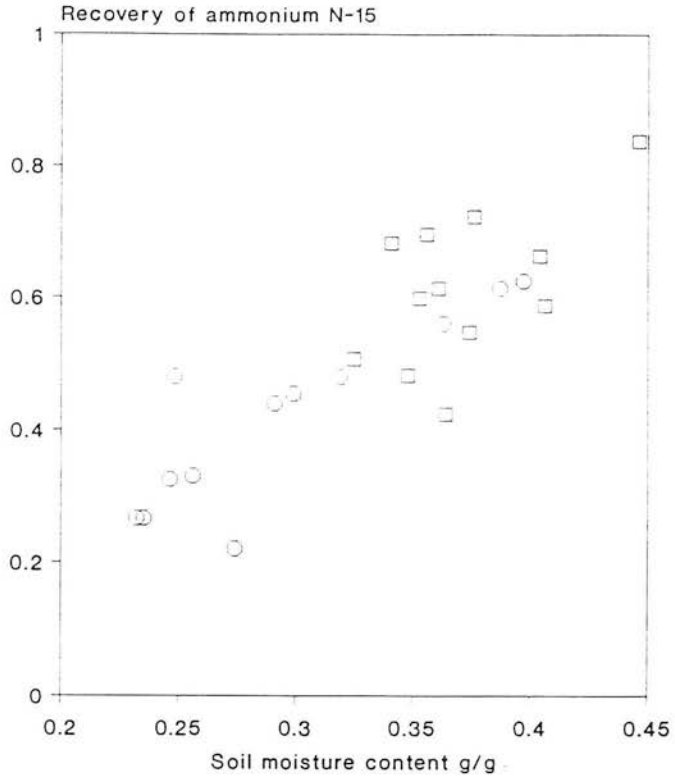

Figure 1. The recovery factor 15 minutes after injection plotted against soil moisture content (g g^{-1}) for two sites: Glencorse (clay loam) (Graph symbol, o) and Beechgrove (sandy loam) (Graph symbol, □). r = 0.854 df = 22.

Table 3. Mean recovery factors for $^{15}NH_4^+$ in the three soils, measured after 15 minutes and 24 hours. 18°C was room temperature.

Site	Time (hours)				
	0.25			24	
	Temperature (°C)				
	18	4	10	14	18
Beechgrove	0.61	0.076	0.054	0.13	N.D.
Glencorse	0.37	N.D.	N.D.	N.D.	0.17
No. 3	N.D.	N.D.	N.D.	N.D.	0.34

N.D. = not determined.

Recovery measurements made within 24 hours of injection for No. 3 field and the Glencorse site showed that the recovery in No. 3 soil was higher than that in Glencorse soil (Table 3). Using the data from extractions of the NH_4^+ pool made on the Beechgrove soils after 24 hours incubation, compared to the amounts of ^{15}N injected, very low 24 hour recoveries were calculated, which were not significantly different between incubation temperatures (Drury and Beauchamp, 1991).

Fixation of NH_4^+ by clays (Drury and Beauchamp, 1991) or organic matter (Foster *et al.*, 1985) are possible mechanisms to explain the rapid removal of $^{15}NH_4^+$ from the extractable pool after addition. Davidson *et al.* (1991) did not observe any significant effect of sterilization on the process and suggested that fixation by vermiculite and other 2:1 clays is the cause. Shen *et al.* (1984) observed that immediately after addition of $^{15}NH_4^+$, 1-5 % of the labelled N was found in the "fixed fraction", extracted by hypobromite.

It is assumed that after adjustment of fixed NH_4^+, equilibriation of the added and exchangeable NH_4^+ rapidly occurs to give a homogeneous exchangeable NH_4^+ pool, which can be simply described in terms of its size and ^{15}N enrichment. The injection methodology should therefore be optimized to give as close to uniform distribution of added ^{15}N as is possible. However, any injection procedure will introduce liquid preferentially into the macropores. Thus the injected NH_4^+ is spatially separated from much of the natural solution, exchangeable and fixed NH_4^+. Given the low effective diffusion coefficient for NH_4^+ in soils (Barber, 1984), it has been calculated that on average in a moist soil NH_4^+ or K^+ will diffuse about 0.13 cm in a day (Wild, 1981). The complex spatial distribution of production, consumption and fixation sites for NH_4^+ (Drury *et al.*, 1991) means that $^{15}NH_4^+$ may well not reach many of the microsites during the initial incubation period and hence it would be liable to a different consumption rate than the natural NH_4^+. If preferential consumption of $^{15}NH_4^+$ is occurring shortly after injection, then overestimates of gross mineralization rates will be made, if that period is included in the calculations.

Davidson *et al.* (1991) observed that there was no difference in the amount of ^{15}N extracted from the sterilized soils at 15 minutes or 24 hours, and therefore suggested that the abiotic reaction is completed very quickly. However, this is contrary to the results of Drury and Beauchamp (1991), who observed that fixation of $^{15}NH_4^+$ continued for at least 3 days. Schimel *et al.* (1989) measured similar consumption rates in intact and mixed cores and suggest that this indicates that the distribution of $^{15}NH_4^+$ was adequately uniform in the intact cores. Although it is not clear what processes are contributing to the rapid fall in the extractable pool of $^{15}NH_4^+$, it would seem more appropriate to allow at least 24 hours for the soil and added NH_4^+ to come into equilibrium before an initial measurement of the ^{15}N content of the NH_4^+ pool is made (Barraclough, pers. comm.). However, other workers only incubate cores for a 24-26 hour period to measure mineralization rates (Davidson *et al.*, 1991; Ambus *et al.*, 1992). Where preliminary tests are carried out on a soil then the addition of ^{15}N can be increased to allow for the $^{15}NH_4^+$ likely to be 'lost' in the first 24 hours, so that the enrichment of the pool does not approach background too quickly. Further work needs to be done to assess the speed with which an injected solution will diffuse throughout the soil pore system in soils of different structures and textures, perhaps using fluorescent dyes. Diffusion models (Darrah *et al.*, 1983) may also be used to assess how rapidly equilibrium between the pools of added and soil NH_4^+ may be attained.

Remineralization

One of the assumptions necessary for the use of most analytical solutions of pool dilution measurements is that enriched NH_4^+ immobilized in microbial tissue during the incubation is not remineralized (Kirkham and Bartholomew, 1954; Nishio *et al.*, 1985). An increase in the size of the ^{15}N content of the NH_4^+ pool strongly suggests that remineralization is occurring (Bjarnason, 1988) and if this is not taken into account in the mathematical framework used, negative gross immobilization rates may result (McTaggart, 1992).

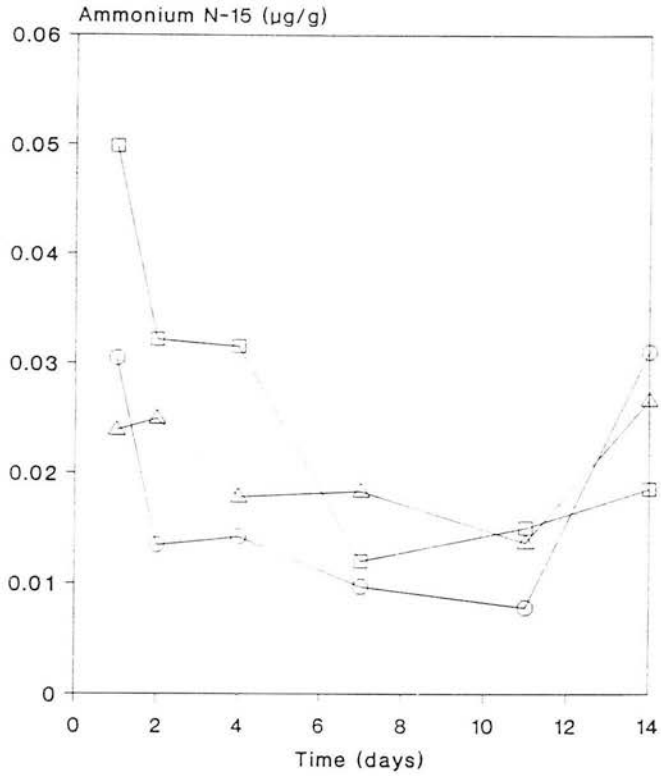

Figure 2. The change in $^{15}NH_4^+$ pool size (µg N g^{-1}) with length of incubation using soil from the Beechgrove site at three incubation temperatures: 4 (o), 10 (Δ), and 14 °C (□).

The incubations of Beechgrove soil cores showed an increase in the ^{15}N content of the NH_4^+ pool after seven days at 14 °C (Figure 2) , whilst at the lower temperatures, where slower turnover might be expected, evidence of remineralization appeared later in the incubations. Incubations should not therefore be carried out for longer than a week (Bristow *et al.*, 1987; Bjarnason, 1988), although at soil temperatures greater than 14 °C, this period might need to be further reduced to minimize the effect of remineralization. Remineralization is likely to be occurring before it is indicated by the increase in size of the $^{15}NH_4^+$ pool and will therefore decrease the decline in ^{15}N abundance and lead to underestimates of the gross rate of mineralization (Wessel and Tietema, 1992). Only by using models which take account of remineralization (mostly numerical) throughout the incubations (Myrold and Tiedje, 1986; Bjarnason, 1988) can gross rates of mineralization be found, where turnover rates are high and remineralization is significant from early in the incubations.

Release of $^{15}NH_4^+$, which has disappeared early in the experiment, can also lead to underestimates of the gross rates of mineralization. Fixed NH_4^+ is only released very slowly and when the solution activity of NH_4^+ is very low (Pasricha, 1976). Shen *et al.* (1984) observed that $^{15}NH_4^+$ fixed immediately after addition was released slowly during an incubation of 20 days with unfumigated soils, but remained constant or increased slightly when previously fumigated soils were incubated. In short-term experiments release of NH_4^+ from sites, where it has been selectively fixed, is unlikely to significantly influence the change in ^{15}N enrichment of the NH_4^+ pool.

Spatial variability of soil mineral nitrogen

Shortly after ploughing at the Beechgrove site in spring, NO_3^- levels were very low and at the bottom of the detectable range. NH_4^+ levels were higher, ranging from 1-7.5 μg N g^{-1} and showing a log-normal distribution (Macduff and White, 1984; Figure 3). The geometric mean was 2.52 μg N g^{-1} and the coefficient of variability was 60 %. Random groups of ten cores selected from the 100 cores sampled indicated that the mean was estimated within its true 95 % confidence interval on 6 out of 10 occasions (Figure 4). The problem of spatial variability of the inorganic nitrogen pool is well known and large samples need to be taken to allow the population mean to be accurately estimated, at this site 40 cores. The sampling of a large number of cores at the beginning and end of the field incubations would allow better estimation of the size of the mineral nitrogen pool, though if cores were covered to prevent leaching the sampling area would also have to be covered.

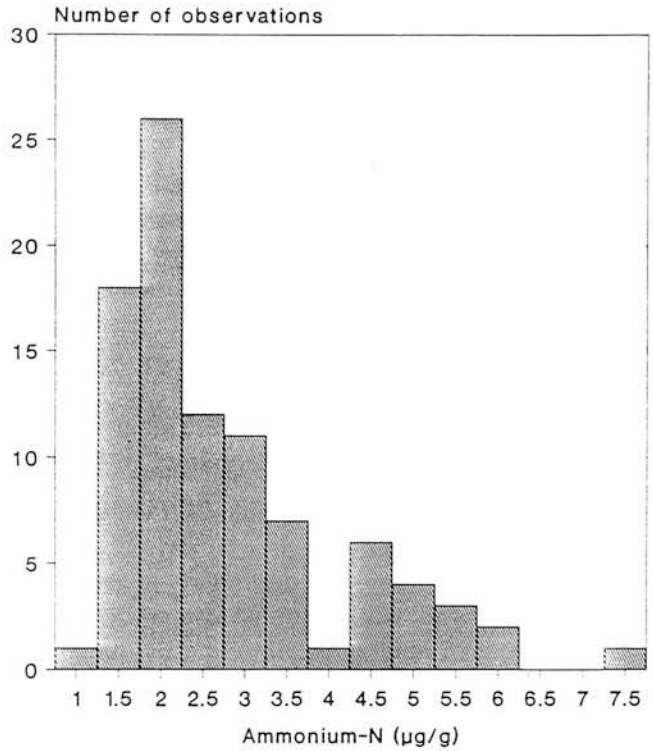

Figure 3. The frequency distribution for NH_4^+ concentration (μg N g^{-1}) determined for 100 cores from the Beechgrove site after spring ploughing.

This study of spatial variability was carried out after the core incubations were completed and indicated that the number of replicates in the experiment was far too low to allow reliable estimation of the mineral N contents of the population of cores at any sampling time. Wessel and Tietema (1992) carried out a separate experiment with a larger number of replicates to follow the changes in the mineral nitrogen pool as a part of their pool dilution experiments. However, practical considerations: initial core sampling;

injection time; available space for the incubation tanks; and analysis of mineral N in the cores; limited the number of replicates that could be included in the core incubations to three injected cores and four cores for assessment of net mineralization. This still resulted in 21 extractions, with 1 l of KCl per extraction, being carried out on each of the sampling days, which was approaching the maximum capacity of the laboratory at that time.

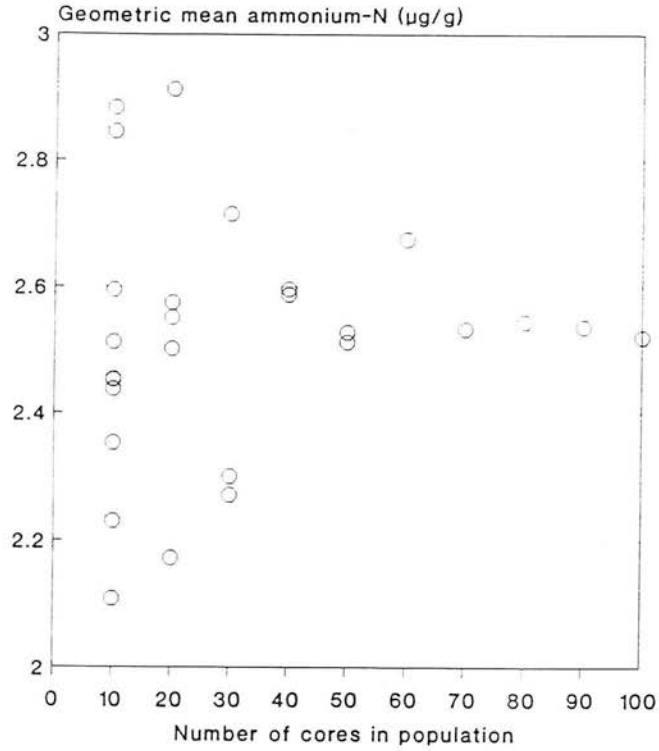

Figure 4. The geometric means for populations of different numbers of cores selected randomly from the 100 cores sampled at the Beechgrove site after spring ploughing against number of cores in the population.

Spatial variability of the soil mineral nitrogen pool and rates of transformation processes (Drury *et al.*, 1991) will lead to spatial variability of the ^{15}N enrichment of the NH_4^+ pool, even where the application is uniform (Davidson *et al.*, 1991). Where such variability is random, small but non-significant errors are introduced to the mineralization rates calculated, though spatially biased distribution of injected $^{15}NH_4^+$, e.g. with respect to depth, should be avoided. Davidson *et al.* (1991) showed by simulation that where less than 70% of mineralizing-immobilizing microsites received ^{15}N as a result of the injection, mineralization rates would be significantly underestimated.

Net mineralization rates

There were no significant differences in NO_3^- and NH_4^+ concentrations in injected and uninjected cores, indicating that injections of small amounts of high enrichment NH_4^+ solution in the experiment did affect rates of net mineralization. This is in accordance with

previous studies showing that net mineralization was unaffected or reduced following additions of inorganic nitrogen (Shen *et al.*, 1984). It was considered by Jenkinson *et al.* (1985) that only in exceptional circumstances (e.g. following recent additions of high C:N ratio residues or when pH is affected by fertilization) would addition of nitrogen affect net mineralization. In the Beechgrove soil, seven replicate cores were therefore used to give the NO_3^- and NH_4^+ concentrations for each temperature, at the end of each incubation period. NH_4^+ and NO_3^- concentrations were widely different between cores and even with seven replicates the standard error of the mean was large (Table 4). This was confirmed by the study carried out in the field after ploughing. In experiments with all the soils, the variability of the NH_4^+ and NO_3^- concentrations within treatments was often more significant than the difference between treatments and this affected the calculation of the net mineralization rates. The net rates of mineralization varied between 3.48 μg N g^{-1} day^{-1} and -1.23 μg N g^{-1} day^{-1}. No significant effects of increasing temperature or moisture on rates of net mineralization were seen, but the No. 3 soil showed more rapid rates of net mineralization (0.25 μg N g^{-1} day^{-1}) than Glencorse (0.13 μg N g^{-1} day^{-1}).

Since estimates of NH_4^+ pool sizes are needed for the calculation of gross mineralization rates, such variability introduces a large degree of uncertainty into the calculation of gross, as well as net, mineralization rates.

Gross mineralization and consumption rates

The estimates of gross mineralization rates in all the soils are similar to those observed by Davidson *et al.* (1991) in grassland soil. Estimates made in forest floor litters give much higher mineralization and consumption rates (Davidson *et al.*, 1991; Wessel and Tietema, 1992). There was no significant effect of increasing moisture potential in the No. 3 soil, with mineralization and consumption rates being estimated at 0.9 μg N g^{-1} day^{-1}. In the Glencorse soil, no significant effect of moisture potential was seen for consumption rates (0.75 μg N g^{-1} day^{-1}). However, mineralization rates increased significantly with decreasing moisture potential from 0.9 μg N g^{-1} day^{-1} at 300 kPa to 1.8 μg N g^{-1} day^{-1} at 10 kPa. In the Beechgrove soil, estimates of gross mineralization rates were made with reference both to day 0 (with a correction for 15 minute recovery) and to day 1 (Table 5). Decreasing rates were seen with increasing incubation period if day 0 was used. However, this trend almost disappeared if concentrations after 1 day were used as the initial conditions. This seems to indicate that it may not be remineralization leading to the fall in gross mineralization rates with time (Wessel and Tietema, 1992) but that some preferential consumption of $^{15}NH_4^+$ is occurring before equilibrium is established. Gross mineralization rates can also be calculated using the previous sampling date to give the new initial values, however, this provides little extra information; the use of measurements from day 1 gives the average of the other combinations. The spatial variability of the soil mineral N pool introduces problems in the calculation of precise gross mineralization rates (as indicated earlier). Mineralization and NH_4^+ consumption rates determined in intact cores indicate a very rapid turnover of the active fraction of the soil organic nitrogen (Ambus *et al.*, 1992).

Table 4. The concentrations of NO_3^- and NH_4^+ in the cores from the Beechgrove site at the end of each incubation period. Values are means of seven replicates with standard errors of the means given in brackets.

Incubation temperature (°C)	Time (days)	NH_4^+ (μg N g^{-1})	NO_3^- (μg N g^{-1})
4	0	0.96 (0.19)	7.05 (1.23)
	1	1.85 (0.26)	9.64 (1.09)
	2	2.01 (0.45)	8.99 (0.89)
	4	3.04 (1.07)	9.34 (0.76)
	7	2.04 (0.27)	10.09 (1.06)
	11	1.81 (0.12)	12.81 (1.00)
	14	5.61 (1.11)	12.45 (1.68)
	18	3.00 (1.04)	11.12 (1.53)
10	0	1.75 (0.18)	5.78 (3.12)
	1	2.60 (0.39)	3.70 (0.53)
	2	2.92 (0.41)	4.59 (1.80)
	4	3.32 (0.71)	3.91 (0.55)
	7	5.14 (0.63)	3.30 (0.48)
	11	3.51 (0.85)	3.46 (0.75)
	14	3.93 (1.45)	10.58 (1.78)
	18	2.62 (0.77)	6.87 (1.85)
14	0	3.51 (1.12)	3.84 (0.78)
	1	4.72 (1.23)	4.24 (1.63)
	2	4.01 (0.93)	5.51 (2.58)
	4	4.35 (1.45)	7.32 (2.78)
	7	2.89 (0.96)	5.00 (0.89)
	11	2.44 (0.68)	4.02 (1.00)
	14	3.25 (1.00)	7.75 (1.57)
	18	2.70 (0.83)	18.38 (3.84)

Table 5. Gross mineralization (m) and consumption (c) rates for the Beechgrove soil for the first seven days calculated with t = 0 and t = 1 as the initial values.

Incubation temperature	Day	m (μg N g^{-1} day^{-1})		c (μg N g^{-1} day^{-1})	
		To t = 0	To t = 1	To t = 0	To t = 1
4 °C	1	4.35		3.67	
	2	3.42	2.91	3.00	2.74
	4	2.82	2.28	2.33	1.85
	7	1.26	0.86	1.13	3.56
10 °C	1	7.30		6.62	
	2	4.01	0.37	3.50	1.89
	4	2.89	1.23	2.53	0.98
	7	2.20	1.20	1.87	0.93
14 °C	1	9.69		8.62	
	2	5.40	2.10	5.22	1.41
	4	2.98	1.01	2.80	1.14
	7	2.52	1.90	2.63	2.20

CONCLUSIONS

The use of pool dilution experiments seems to provide a method for measurement of the gross rates of some of the simultaneous and often opposing processes occurring in soils. However, the soil internal nitrogen cycle model proposed by Jansson (1958) does not always fit all the observed results (Myrold and Tiedje, 1986; Drury *et al.*, 1991). Pool dilution experiments may well pose as many questions as they answer about the interlocking processes occurring in the soil. Use of the method must be accompanied by a check that the underlying assumptions are valid and experiments must be designed to minimize error multiplication factors (Wessel and Tietema, 1992). Most information may be obtained, where the fate of ^{15}N added is also determined at the end of the incubation period.

Optimizing the injection procedure for the soil type on which mineralization measurements are to be carried out can help to ensure that a homogeneous NH_4^+ pool is achieved as rapidly as possible. The injection procedure should be checked for each soil to be studied using dyes and/or modelling. Even where uniform addition of $^{15}NH_4^+$ can be achieved, slow equilibriation of NH_4^+ between the fixed and exchangeable pools, limited by diffusion, will also lead to a pool dilution effect and added $^{15}NH_4^+$ and natural NH_4^+ may be subject to different consumption rates initially. While this problem is exacerbated by non-uniform additions in intact cores, it may also be a problem in laboratory incubations. Short-term recovery analysis to correct for the amount of $^{15}NH_4^+$ becoming irrecoverable is necessary (Davidson *et al.*, 1991). However, from our results it would seem more appropriate to determine the initial NH_4^+ pool size and enrichment at least 24 hours after injection. It is also important that the ^{15}N natural abundance in the soil organic N, or active pool if possible, is also determined at the study site.

Davidson *et al.* (1991) suggest that estimates of rates are most accurate where drops in the enrichment of the pool are large over the incubation period, but the final ^{15}N enrichment should not approach too closely to background (Wessel and Tietema, 1992). Larger additions of $^{15}NH_4^+$ would help to achieve these aims, especially in soils where recovery values are low, but care must be taken as the system may be dramatically altered if substrate is added to a substrate-limited process.

Application of analytical solutions to calculate gross rates means that incubations should be completed before recycling becomes significant. Recycling was observed after seven days in the experiment with the Beechgrove soil. However, remineralization may become significant much earlier. The length of incubations should therefore be limited to approximately one week after injection, where soils temperatures do not exceed 14 °C. This limits the use of analytical solutions to incubations carried out over very short periods, once an initial equilibriation has been carried out. The use of numerical solutions may remove this restriction. However, numerical models may need to be expanded to take NH_4^+ fixation into account.

The measurement of gross mineralization rates can however only produce estimates of these rates especially where measurements are made *in situ*. The spatial variability of mineral nitrogen in soils leads to a measurement of initial and final mineral nitrogen pools with significant errors attached, which are multiplied in the calculation of gross mineralization rates. The use of a parallel experiment using a large number of uninjected cores over the same or a longer time period than the injected cores, allows an increase in the accuracy with which pool sizes can be measured.

The use of pool dilution techniques may on many occasions seem to have more problems than advantages. However, the continued sensible use and continual reassess-

ment of these procedures can lead us further in our understanding of the processes controlling the supply of mineral nitrogen from soils in forms available to plants.

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APPENDIX 7

Nitrogen supply for organic cereal production in Scotland

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33 Nitrogen supply for organic cereal production in Scotland

E. A. Stockdale[1], R. M. Rees[1] and M. G. Davies[2]

Problems in supplying sufficient N to an organic cereal crop to meet the demand for early N uptake are highlighted. Manures containing a greater proportion of N in mineral forms do not seem to give any yield advantage to the crop. However, crop N uptake was closely correlated with the mineral N recovered in the profile after the manure application. This mineral N was recovered more efficiently where poultry manure (rather than farmyard manure) had been applied, and this was possibly associated with higher microbiological activity following poultry manure application. Leaching was reduced where a cover was maintained overwinter; however, the use of rye-grass as a cover crop restricted both yield and N accumulation of a following cereal crop. Leaving stubble overwinter was the most suitable overwinter strategy, but this may cause problems with disease transmission between crops.

Introduction

The supply of nitrogen is one of the principal factors limiting organic crop production. Nitrogen is applied to the soil predominantly in the form of animal manure and crop residues. The release of N in forms available for plant uptake (NH_4^+ and NO_3^-) from complex organic materials is effected by the soil micro-organisms which control the decomposition of organic materials, depending upon the environmental conditions and chemical compounds in the material (Daji, 1934).

There have been many studies investigating the amounts and timing of N released from a range of manures and crop residues under laboratory, greenhouse and field conditions (e.g. Rees *et al.*, 1993; Beauchamp, 1986). The main factors controlling N release have been known at a qualitative level since the 1930s (Daji, 1934), but quantitative prediction of N release has remained difficult (de Willingen, 1991). As well as the release of nitrogen from manures, an important factor is the ability of crops to take up and utilize this N efficiently. The demand of crops for N is not constant, and one of the main challenges to organic crop production is to match the timing of N supply with crop demand, linking it to the sowing date and length of crop season (Stockdale *et al.*, 1992). Where this can be achieved, maximum yields will be obtained with minimum

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losses of N to the environment (Powlson, 1988). For example Stockdale *et al.* (1992), found that in an organic system a potato crop used N released from heap-composted FYM more efficiently than spring barley, despite the higher total N uptake of the potato crop. This was linked to the later sowing and longer period of N demand by the potato crop.

Scotland can be regarded as having an adverse climate for crop production. The growing season is restricted by cold winter temperatures and lower summer temperatures than southern Britain. Cereal growth is generally at a disadvantage, since cereals have a high nutrient demand early in spring when temperatures are low and mineralization rates restricted (Redman *et al.*, 1989). Late harvests also restrict the choice of cover crops that can be used after cereals. Scotland is also significantly wetter than much of southern England; for example, rainfall in Fife is on average 30 per cent greater than that measured at Bedford. However, effective rainfall in winter may be more than double that in southern England, leading to an increased likelihood of NO_3^- leaching.

Three field trials over two seasons in southern Scotland studied the release of N from a range of manures, including heap-composted farmyard manure, poultry manure and grass-clover leys, and the efficiency with which the N was used by a following cereal crop. The risk of N losses overwinter by leaching was also assessed and monitored directly after the ploughing-out of grass-clover leys. Table 1 shows topsoil properties and previous cropping for all three trials.

Methods

Trial 1: Rates and form of manures for spring barley, Jamesfield, 1991

A large trial was carried out to investigate the effect of different manure types and rates on the yield and N uptake pattern of spring barley. The efficiency of use of the inorganic N pool added in the manure was also estimated using ^{15}N. The experimental treatments (no manure and three rates of heap-composted farmyard manure and poultry manure; Table 2) were replicated four times and laid out in a Youden Square design with four blocks of seven plots of 18 x 6 m. Manure was applied by hand to the plot surface on 9 April and this was incorporated by Rotaspike. Drilling with spring barley took place on 18 April 1992. Microplots (2 x 1.5 m) were included in each manured plot, in which the inorganic N pool of the manure was labelled with a solution of $(^{15}NH_4)_2SO_4$, applied with a fine mist sprayer. This gave ^{15}N enrichments in the inorganic N pool of approximately 13 atom per cent for the poultry manure and 8 atom per cent for the farmyard manure.

Soil samples were taken on 23 April in the microplots to assess the actual ^{15}N enrichment in the mineral N in the soil. Anaerobic incubations (Keeney, 1982) and an extraction with hot KCl (McTaggart and Smith, 1993) were also carried out on these soils to determine whether these methods were suitable N availability indices for manured soils. Soil and plant samples in the plots and microplots were taken in conjunction at intervals throughout the

season, and a plot combine was also used to assess the grain yield at the final harvest.

Table 1 Topsoil properties (0-0.3 m) and previous cropping for the cereal trials. Textures are given using the coding: sand (S), clay (C), silt (Z) and loam (L)

	Trial 1	Trial 2	Trial 3
Year	1991	1992	1993
Crop	Spring barley	Oats	Spring barley
Farm	Jamesfield	Jamesfield	Boghall
Town	Abernethy	Abernethy	Penicuik
Grid Ref. No	204182	198178	248653
Soil series	Carpow/Carey	Carey	Macmerry/Duncrahill
Textural group	SCL	S(C)L	SL
Organic matter %	3.1	3.2	8.6
Total N mg kg^{-1}	1500	1600	3400
pH	6.8	6.6	6.3
Previous crop	Potatoes	Spring barley	Grass/clover Spring barley

Table 2 Rates of poultry manure and heap-composted farmyard manure applied to spring barley on April 9th 1992 at Jamesfield Farm (Trial 1)

Treatment	Manure rate (t ha^{-1})	Total N (kg ha^{-1})	NH_4^+-N (kg ha^{-1})
Control	0.0	0.0	0.0
Low FYM	6.5	40.2	1.7
Medium FYM	27.8	172.2	7.2
High FYM	56.5	350.2	14.7
Low PM	2.8	27.8	3.8
Medium PM	13.0	129.6	17.9
High PM	20.4	203.7	28.2

Trial 2: Cover crops trial, Jamesfield, 1991-92

The efficiency of four management strategies in using residual mineral N in the profile was monitored overwinter. The treatments were: autumn ploughed and fallow; autumn ploughed, with rye grass sown; autumn ploughed, with a rye grass/red clover mix sown, and stubble left overwinter. Four replicate plots (20 x 6 m) of each treatment were laid out in a Latin Square design in October 1991. Microplots (2.5 x 2 m) in each treatment received $(^{15}NH_4)_2SO_4$ (99.2 atom per cent) (applied using a fine mist sprayer), at a rate of 1.66 kg N ha^{-1} to label the residual profile mineral N. Soil samples were taken after application of the ^{15}N, to allow initial enrichments of the profile residual N to be calculated.

Trial cover crops were ploughed out on 11 March 1992, after a sample had been taken to determine the yield and ^{15}N enrichment of the cover crop in each treatment and then drilled with oats in early April. The N uptake pattern of the oats and the efficiency of N use contained in the cover crops was monitored by sequential sampling of plant and soil. Grain yield at final harvest was also established.

Trial 3: Ploughed-out leys, Bush, 1992

Three old ley trial plots were ploughed out in late February: a five-year grass-clover ley; a one-year grass-clover ley; and a five-year rye-grass ley, and these were compared in the trial with a plot under continuous arable cultivation. Spring barley was drilled on 25 March and soil and plant samples were taken at intervals to monitor the yield and N uptake of the spring barley crop. After harvest, soil sampling continued and three porous cup samplers were installed at 0.4 m on 20 October in both the plots and the grass-clover guard area to provide a control. Weekly water samples were withdrawn from the cups following an application of 20 kPa suction for 48 hours, to allow winter leachate concentrations to be assessed. Drainage data from hydrologically isolated plots on the Bush estate (Vinten *et al.*, 1992) were used to calculate leaching losses.

Laboratory analysis

Plant samples, cut to within a few mm of ground level, were oven dried at 100°C and progressively milled to produce a very fine flour. Samples were then analysed for total N and ^{15}N enrichment. Soil samples were taken in the metre-square area from which the plants had been removed for estimation of dry matter. Three cores were taken to a depth of 0.3-0.35 m and sealed in plastic bags to prevent moisture loss. Samples were stored at 5°C overnight, where analysis was to occur the next day. Otherwise they were stored frozen (-15°C) until analysis could be carried out. After sieving, available ammonium and nitrate were extracted from the soil using 1 M KCl. The extractant was filtered and NH_4^+-N and $NO3^-$-N determined by continous flow analysis. Extracts were prepared for ^{15}N analysis, where appropriate, by steam distillation.

Results and discussion

Early mineral N for cereals

Poultry manure contains significantly more N in available forms (dominantly as NH_4^+ and easily hydrolysed uric acid compounds) than the heap-composted farmyard manure (Hadas *et al.*, 1983). Very highly significant differences in NO_3^- concentration were observed between treatments after drilling in Trial 1 (Figure 1), which were strongly correlated with the amount of N added as NH_4^+ ($r = 0.8$). These differences persisted until early July, when NO_3^- concentrations in the soil fell to a very low level, mirroring crop N uptake.

Grain yields during the three trial years were similar to those obtained from other organic farms in Scotland, and the N concentration of the barley grain did not exceed the 1.7 per cent ceiling for malting (Table 3). Table 4 shows N uptake by the spring barley was not significantly affected by the form of manure used, but significant differences ($p < 0.05$) were seen between rates of manure application. The low rates of manure application did not increase the N uptake of the barley above the control. The plots, which had received medium and high rates of manure application, showed increased N uptake, although the difference between the highest measured N uptake and the control was usually the only

significant difference. Recovery of the N from the whole manure was estimated by the difference method (i.e. the difference in N uptake between manured and non-manured plots) and ranged from 5 to 21 per cent at harvest. Recovery of N from the poultry manure was significantly higher on average (16.1 per cent) than that from the farmyard manure (7.5 per cent), and only a little higher than the percentage of manure N present as mineral N, 14 per cent for the poultry manure and 4 per cent for the farmyard manure. Recoveries were significantly lower for the high rate of application ($p < 0.05$).

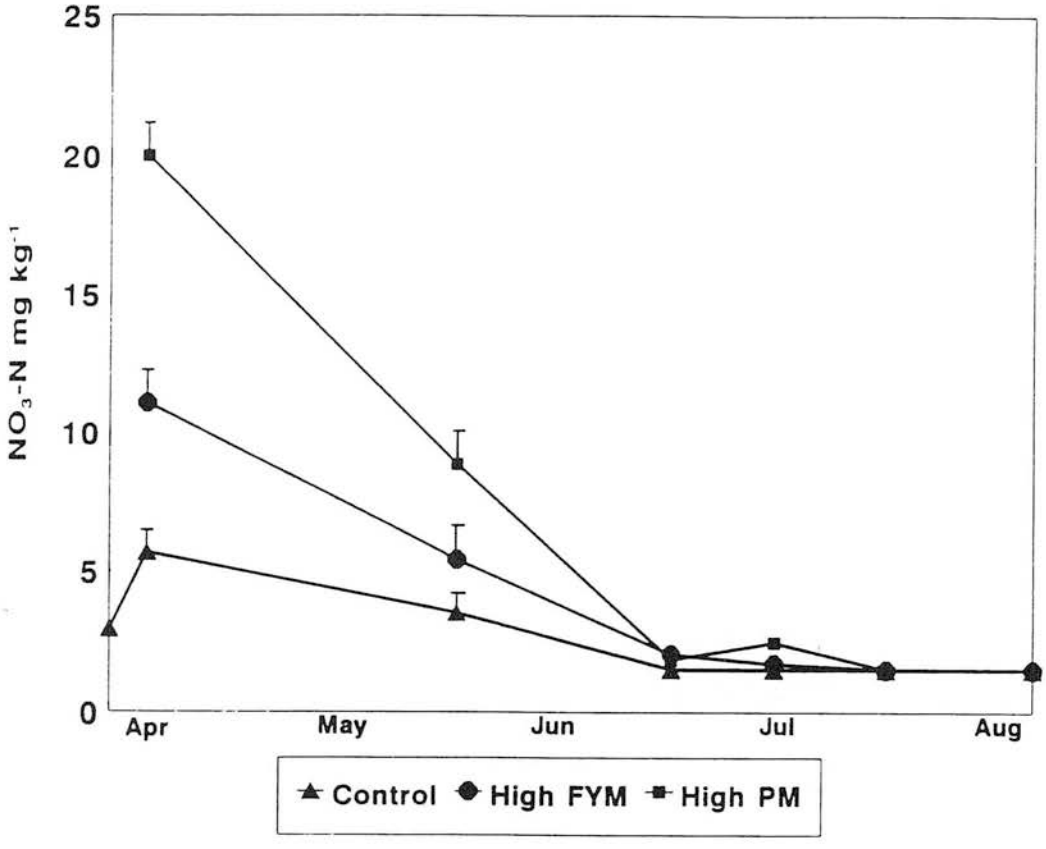
Table 3 Grain yields (t ha^{-1}) @ 85 % dry matter and grain N concentrations (%) for the three cereal trials 1991-2. Values given are means of four replicate plots

	Crop	Yield (t ha^{-1} @ 85% DM)	N concentration (%)
Trial 1 1991	Spring barley		
Control		2.4	1.3
Low-rate poultry manure		2.2	1.3
Medium-rate poultry manure		3.1	1.4
High-rate poultry manure		3.1	1.4
Low-rate farmyard manure		2.0	1.2
Medium-rate farmyard manure		2.8	1.4
High-rate farmyard manure		3.1	1.4
Trial 2 1992	Oats		
Bare fallow		4.6	1.5
Stubble fallow		4.7	1.6
Rye-grass		3.9	1.3
Rye/clover		3.9	1.3
Trial 3 1992	Spring barley		
5 yr grass/clover		5.4	1.3
1 yr grass/clover		4.2	1.3
5 yr grass		5.0	1.4
Continuous arable		2.7	1.3

Table 4 N uptake of spring barley in Trial 1, kg ha^{-1}, after application of 3-rates of poultry and farmyard manure. Values are means of four replicate plots for each treatment, with the least significant difference between the means for each harvest given

Treatment	Cut 1	Cut 2	Cut 3	Cut 4
Control	13.48	25.17	24.37	32.33
Poultry manure				
Low	12.90	27.61	29.00	37.26
Medium	33.00	41.89	51.33	59.45
High	29.21	46.17	49.41	52.17
Farmyard manure				
Low	13.24	27.20	30.48	34.36
Medium	27.68	42.39	45.14	53.71
High	29.10	41.91	44.04	49.94
LSD	18.676	18.439	19.468	22.104

Figure 1 **Nitrate concentration in the topsoil (0-0.3 m), where three-rates of poultry and farmyard manure had been applied to the soil on April 9 (Day 99) Trial 1. See Table 2 for treatments. PM = Poultry Manure, FYM = Farmyard Manure**



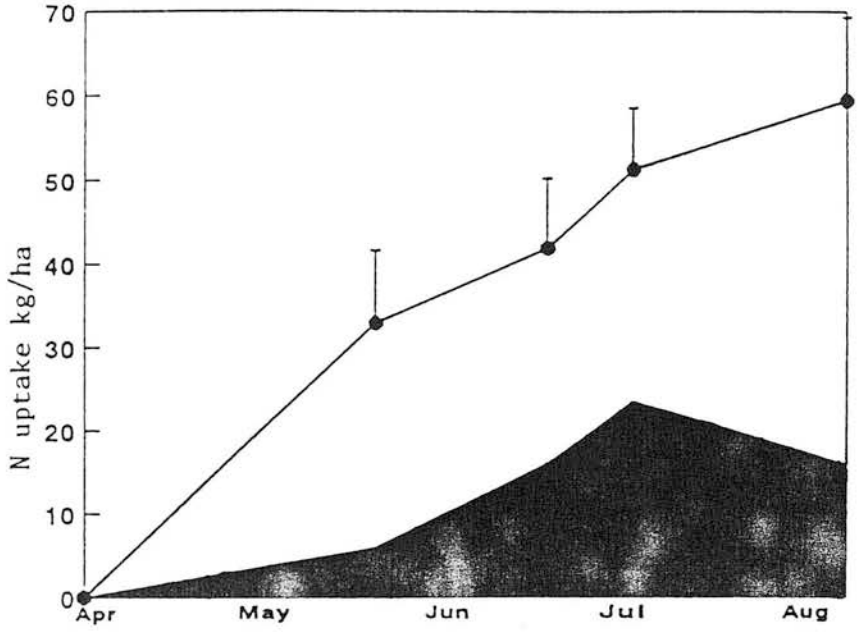
Recovery of the ^{15}N-labelled mineral N pool of the manure was significantly different between manure types ($p < 0.01$), with greatest recovery (103 per cent) from the poultry manure. There were no significant differences between the rates of poultry manure application, but much greater recoveries (approximately 100 per cent) were measured at the low rate of farmyard manure application, possibly reflecting an increased efficiency of recovery, than at the higher rates (65.9 per cent at the third harvest).

Uptake of the N from the mineral-N added in the manure was between 6 to 42 per cent of total plant uptake at harvest (Figure 2). The proportion of plant-N derived from this pool was significantly higher in the plots treated with poultry manure than those receiving farmyard manure ($p < 0.01$), and there was some evidence at harvest, though not earlier, that the proportion of plant uptake increased with increasing addition of mineral ^{15}N. A stepwise regression procedure, including all possible factors, identified NO_3^- measured after drilling as the only significant factor, which explained differences in N uptake by the barley crop (Figure 3). This was probably a better predictor than applied NH_4^+,

since it allowed for losses and fixation of ammonium during and after application.

Figure 2 **N uptake of a spring barley crop (Trial 1) treated with: a) medium-rate of poultry manure and b) medium-rate of farmyard manure. In each case the total N uptake is represented by the line through the mean of four replicate plots, standard errors of each mean are indicated by the bar.**

a) Medium rate poultry manure



b) Medium rate farmyard manure

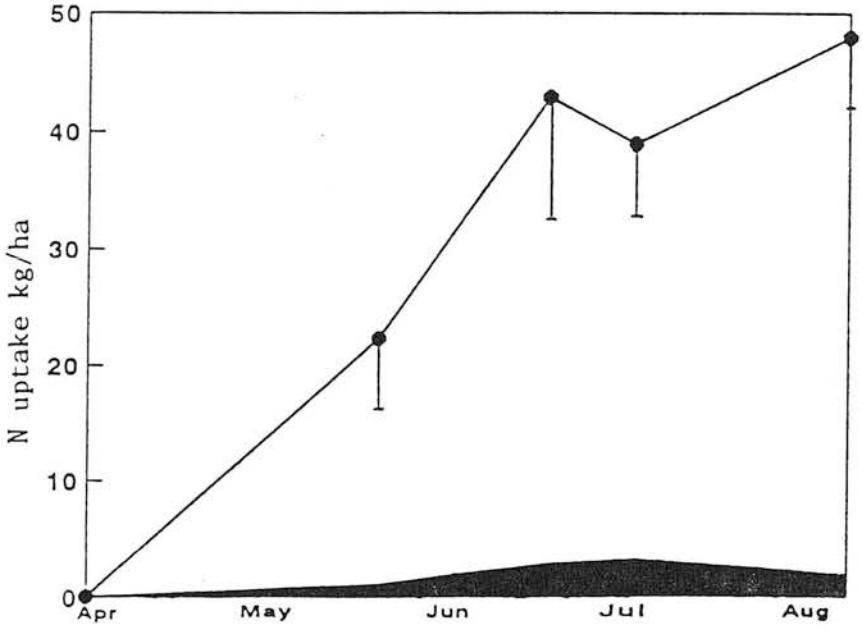
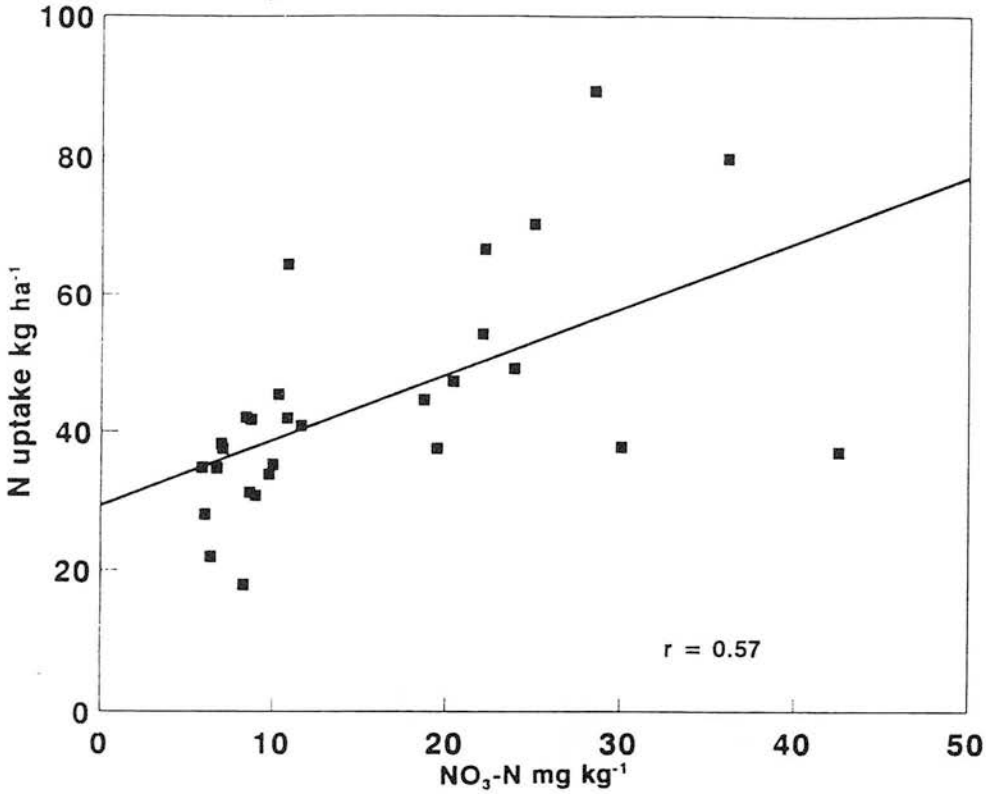


Figure 3 **N uptake of spring barley (Trial 1) at harvest plotted against mineral N present in the soil approximately 2 weeks after manure application and drilling**



It has been proposed that ammonium-N added in the manure is the most important factor controlling the N uptake of the crop (Beauchamp, 1986). However, where large amounts of soluble C are added in the manure, the incorporation of the added mineral-N into the soil biomass seem to have rendered it more available for plant uptake over the season than mineral-N supplied in conventional fertilizer. This may be due to the prevention of losses of N before the crop roots are fully developed, or that N released as a result of microbial death or predation is more available for crop uptake.

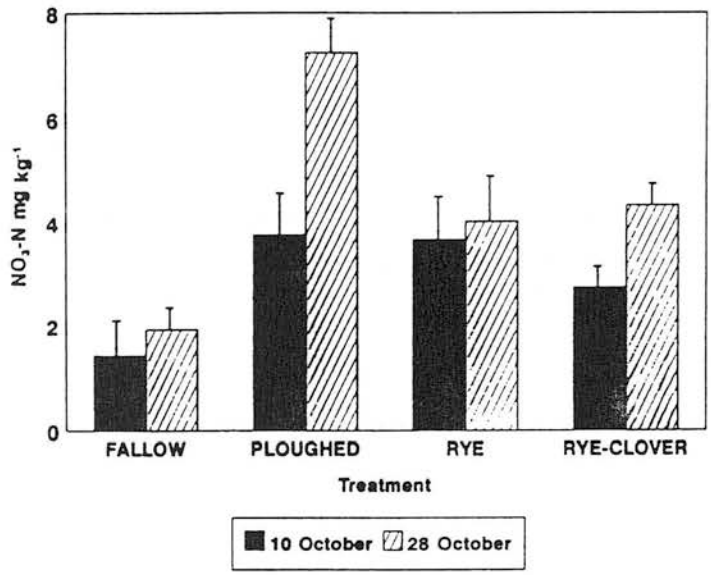
The incorporation of leys (Trial 3) seemed to provide a large supply of N available for uptake by the spring barley crop, and this trial gave the largest yields of spring barley (Table 3). Dry matter yields following the incorporation of any of the previous leys were very significantly higher ($p < 0.01$) than those from the continued arable plots. As expected, the barley crop following the oldest ley (where most organic matter had accumulated) showed the highest increase in yield.

Overwinter management and leaching of NO_3^-

The ploughing of the stubble after harvest stimulates the mineralization of crop residues and soil organic matter (Colbourn, 1985). Increases in NO_3^- (Trial 2) were reduced where a cover crop had been sown after ploughing or the plots had been left in stubble fallow (Figure 4). Leaching losses of residual NO_3^- are therefore thought to have been reduced in the presence of any active cover

overwinter (Atallah and Lopez-Real, 1991). Rye-grass and the rye-grass/clover mixture yielded approximately 1 t ha^{-1} dry matter between October and March, with little clover establishment seen overwinter, while little weed growth occurred on the bare fallow plots (Trial 2). However, the stubble fallow treatment produced the largest dry matter yield overwinter (4 t ha^{-1}) and showed the largest grain N uptake of 75 kg ha^{-1}, compared to 69 kg ha^{-1} in the bare fallow treatment. The plots with a cover crop or stubble fallow overwinter recovered 24 per cent of the residual profile mineral N, with no significant differences between these treatments. However, the ploughed fallow plot recovered significantly less of the residual profile mineral N (2 per cent) than any of the plots with an overwinter cover ($p < 0.05$).

Figure 4 **NO_3^- concentration in the soil (0-30 cm) in Trial 2 two weeks after ploughing (10 October) and four weeks after ploughing (28 October), when the cover crops had emerged**



Estimated leaching losses following the ploughing out of leys have varied widely (Ryden *et al.*, 1984) and are related to the previous season's management and the age of the ley (Whitehead *et al.*, 1990). Although the ploughed-out five-year grass-clover ley resulted in the highest losses of N by leaching (Trial 3; Table 5) they were not significantly greater than those from the continued ungrazed grass-clover sward. Since porous cups were installed after the onset of drainage, these estimates (Table 5) must be considered as underestimates, though data from soil cores and porous cups suggest that the majority of nitrate loss was accounted for. Organic farming systems rule out the use of herbicide and therefore strong regrowth occurred beneath the barley crop following the ploughed-out leys. Since the plots were not ploughed in the autumn, regrowth provided a substantial sink for nitrate overwinter and therefore potential leaching from these plots was reduced. This may account for the lower

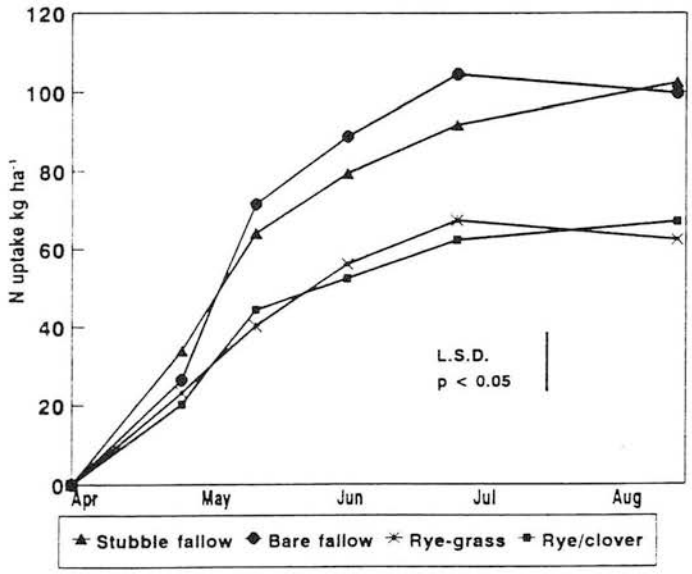
losses of N after the ploughed-out grass compared to the plot under continued arable cultivation, where there was little soil cover overwinter.

Table 5 Mean estimated leaching losses for the period 16 October 1992 to 22 February 1993 for various treatments. Standard errors of the means (SE) are also given.

Treatment	Mean leaching loss (kg N ha^{-1})	SE
Continued grass-clover ley	3.1	0.40
5 year grass-clover ley	12.8	4.45
5 year rye-grass ley	6.1	1.40
Continued arable cultivation	10.1	3.12

The use of rye-grass or a rye-clover mix seemed to restrict the N uptake of the following oat crop (Figure 5), with significant differences between plots which had remained fallow (bare or stubble) and plots which had had a rye or rye/clover cover. Nitrate concentrations in the crop plots were significantly lower than the concentrations in the stubble plots for six weeks after incorporation ($p < 0.05$). This was thought to be because the C:N ratio of the rye-grass was sufficiently high to immobilize significant quantities of mineral N.

Figure 5 Mean N uptake (kg ha^{-1}) by an oat crop (Trial 2) following various overwinter treatments: bare fallow; stubble fallow; rye-grass and rye/clover cover crop. Each point is the mean of four replicate plots, with the S.E. of the mean indicated by a bar.



Stubble fallowing seems to be the most appropriate, and cheapest, over-winter management technique before a spring cereal crop, although such practices can lead to the transmission of disease (e.g. Take all) between cereal crops. Forage rape may be an alternative, but in situations where the leaching of NO_3^- is not perceived to be important, a winter bare fallow might be the better option.

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